Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

Lower Duwamish Waterway Remedial Investigation

REMEDIAL INVESTIGATION REPORT

FINAL

For submittal to:

The US Environmental Protection Agency
Region 10
Seattle, WA

The Washington State Department of Ecology
Northwest Field Office
Bellevue, WA

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Prepared by: Windward environmental LLC

200 West Mercer Street, Suite 401 • Seattle, Washington • 98119

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Acronyms

Acronym	Definition
ac	acres
AET	apparent effects threshold
AOC	Administrative Order on Consent
AOC	area of concern
API	Asian and Pacific Islander
ARI	Analytical Resources, Inc.
aRPD	apparent redox potential discontinuity
AS/SVE	air sparging/soil vapor extraction
ASTM	American Society for Testing and Materials
ATSDR	Agency for Toxic Substance and Disease Registry
AWQC	ambient water quality criteria
BACT	best available control technology
Battelle	Battelle Marine Research Laboratory
BAZ	biologically active zone
BBP	butyl benzyl phthalate
BCA	bias-corrected accelerated
BDC	Boeing Developmental Center
BEHP	bis(2-ethylhexyl) phthalate
bgs	below ground surface
BHC	benzene hexachloride
BMP	best management practice
BOD	biological oxygen demand
Boeing	The Boeing Company
BTEX	benzene, toluene, ethylbenzene, and xylene
CAA	Clean Air Act
CAS	Columbia Analytical Services, Inc.
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CERCLIS	Comprehensive Environmental Response, Compensation, and Liability Information System
CFR	Code of Federal Regulations
cfs	cubic feet per second
COC	chemical of concern
COPC	chemical of potential concern
CORRACTS	Corrective Action Site
cPAH	carcinogenic polycyclic aromatic hydrocarbon

Acronym	Definition
CPM	coarse particulate matter
^{137}Cs	cesium-137
CSCSL	Confirmed and Suspected Contaminated Sites List
CSL	cleanup screening level
CSM	conceptual site model
CSO	combined sewer overflow
CT	central tendency
CVAF	cold vapor atomic fluorescence
CWA	Clean Water Act
cy	cubic yard
DAIS	Dredged Analysis Information System
DARRP	Damage Assessment, Remediation, and Restoration Program
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
DEA	David Evans and Associates, Inc.
DMMO	Dredged Material Management Office
DMMP	Dredged Material Management Program
DMMU	dredged material management unit
DNAPL	dense non-aqueous phase liquid
DO	dissolved oxygen
DPD	Department of Planning and Development
DQO	data quality objective
DRCC	Duwamish River Cleanup Coalition
DSOA	Duwamish sediment other area
dw	dry weight
EAA	early action area
EBI	Elliott Bay Interceptor
Ecology	Washington State Department of Ecology
ECOSS	Environmental Commission of South Seattle
EDR	Environmental Data Resource, Inc.
EE/CA	engineering evaluation/cost analysis
EFDC	Environmental Fluid Dynamics Code
EIM	Environmental Information Management
EOF	emergency overflow
EPA	US Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment

Acronym	Definition
ERNS	Emergency Response Notification System
ERTS	Environmental Report Tracking System
ESA	Endangered Species Act
ESG	Environmental Solutions Group, Inc.
FPM	fine particulate matter
FR	Federal Register
FS	feasibility study
FWM	food web model
GC/ECD	gas chromatography with electron capture detection
GC/MS	gas chromatography/mass spectrometry
GIS	geographic information system
gpd	gallons per day
GPS	global positioning system
GSA	General Services Administration
GSD	grain size distribution
GWI	Great Western International
Herrera	Herrera Environmental Consultants
HHRA	human health risk assessment
HI	hazard index
HOC	hydrophobic organic compound
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HpCDD	heptachlorodibenzo-*p*-dioxin
HpCDF	heptachlorodibenzofuran
HPLC/PDA	high-performance liquid chromatography/photodiode array detection
HQ	hazard quotient
HSD	honestly significant difference
HSL	Hazardous Sites List
HWMP	Hazardous Waste Management Program
HxCDD	hexachlorodibenzo-*p*-dioxin
HxCDF	hexachlorodibenzofuran
I-5	Interstate 5
ICP-MS	inductively coupled plasma-mass spectrometry
ID	identification
IDW	inverse distance weighting
IIT	Illinois Institute of Technology
IP	intraperitoneal
IQR	interquartile range

Acronym	Definition
IR	ingestion rate
IRA	independent remedial action
IWP	Industrial Waste Program
J-qualifier	estimated concentration
JN-qualifier	tentative identification of estimated concentration
KCDNR	King County Department of Natural Resources
KCIA	King County International Airport
K-M	Kaplan-Meier
LAET	lowest apparent effects threshold
2LAET	second lowest apparent effects threshold
LDW	Lower Duwamish Waterway
LDWG	Lower Duwamish Waterway Group
LF	linear feet
LNAPL	light non-aqueous phase liquids
LOAEL	lowest-observed-adverse-effect level
LOEC	lowest-observed-effect concentration
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
LQG	large-quantity generator
LUST	leaking underground storage tank
M-qualifier	calculated concentration
MDL	method detection limit
mgd	million gallons per day
mgy	million gallons per year
ML	maximum level
MLLW	mean lower low water
MNR	monitored natural recovery
MT/yr	metric tons per year
MTCA	Model Toxics Control Act
N-qualifier	tentative identification
NAPL	non-aqueous phase liquid
NBF	North Boeing Field
NFA	No Further Action
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
NPDES	National Pollutant Discharge Elimination System
NPL	National Priorities List

Acronym	Definition
NTCRA	non-time-critical removal action
NTR	National Toxics Rule
NWR	National Wildlife Refuge
O&M	operation and maintenance
OC	organic carbon
OCDD	octachlorodibenzo-*p*-dioxin
OCDF	octachlorodibenzofuran
OIG	Office of the Inspector General
OSWER	Office of Solid Waste and Emergency Response
Pa	pascal
PAH	polycyclic aromatic hydrocarbon
^{210}Pb	lead-210
PCB	polychlorinated biphenyl
PCE	tetrachloroethene
pcf	pounds per cubic foot
PCP	pentachlorophenol
PCT	polychlorinated terphenyl
PeCDD	pentachlorodibenzo-*p*-dioxin
PeCDF	pentachlorodibenzofuran
PEF	potency equivalency factor
PIE	Pacific International Engineering, PLLC
PM	particulate matter
PM$_{10}$	particulate matter less than 10 micrometers in diameter
PM$_{2.5}$	particulate matter less than 2.5 micrometers in diameter
ppb	parts per billion
ppt	parts per thousand
PRG	preliminary remediation goal
PSAMP	Puget Sound Ambient Monitoring Program
PSC	Phillip Services Corporation
PSCAA	Puget Sound Clean Air Agency
PSDDA	Puget Sound Dredged Disposal Analysis
PSEP	Puget Sound Estuary Program
PSMAF	Puget Sound Maritime Air Forum
PVC	polyvinyl chloride
QA/QC	quality assurance/quality control
QAPP	quality assurance project plan

Acronym	Definition
QEA	Quantitative Environmental Analysis, LLC
R^2	regression coefficient
RAO	remedial action objective
RBC	risk-based concentration
RBTC	risk-based threshold concentration
RCRA	Resource Conservation and Recovery Act
RCW	Revised Code of Washington
RfD	reference dose
RFI	RCRA facility investigation
RI	remedial investigation
RL	reporting limit
RM	river mile
RME	reasonable maximum exposure
ROC	receptor of concern
ROD	Record of Decision
ROW	right-of-way
RPS	relative penis size
PSS	practical salinity scale
SAIC	Science Applications International Corporation
SARA	Superfund Amendments and Reauthorization Act
SCA	source control area
SCAP	source control action plan
SCWG	Source Control Work Group
SD	storm drain
SEA	Striplin Environmental Associates, Inc.
SEPA	State Environmental Policy Act
SF	slope factor
SI	site inspection
SL	screening level
SMC	Seattle Municipal Code
SMS	Washington State Sediment Management Standards
SPCC	South Park Community Center
SPI	sediment profile imaging
SPMD	semipermeable membrane device
SPU	Seattle Public Utilities
SPWG	Sediment Phthalates Work Group
SQG	small-quantity generator
SQS	sediment quality standards

Acronym	Definition
SRI	supplemental remedial investigation
STAR	sediment transport analysis report
STM	sediment transport model
SU	standard unit
SVOC	semivolatile organic compound
SWAC	spatially weighted average concentration
SWF/LF	solid waste facility/landfill
SWMU	solid waste management unit
T-105	Terminal 105
T-106	Terminal 106
T-107	Terminal 107
T-108	Terminal 108
T-115	Terminal 115
T-117	Terminal 117
TBD	to be determined
TBT	tributyltin
TCDD	tetrachlorodibenzo-*p*-dioxin
TCDF	tetrachlorodibenzofuran
TCE	trichloroethene
TCRA	time-critical removal action
TE	trapping efficiency
TEF	toxic equivalency factor
TEQ	toxic equivalent
TOC	total organic carbon
TPH	total petroleum hydrocarbons
tpy	tons per year
TRIS	Toxics Release Inventory System
TRV	toxicity reference value
TSD	treatment, storage, and disposal
TSS	total suspended solids
U&A	Usual and Accustomed
U-qualifier	not detected at given concentration
U.S.C.	United States Code
UCL	upper confidence limit
USACE	US Army Corps of Engineers
USCG	United States Coast Guard
USFWS	US Fish and Wildlife Service
USGS	US Geological Survey

Acronym	Definition
UST	underground storage tank
VCP	Voluntary Cleanup Program
VOC	volatile organic compound
WAC	Washington Administrative Code
WARM	Washington Ranking Method
WDFW	Washington State Department of Fish and Wildlife
WHO	World Health Organization
Windward	Windward Environmental LLC
WPCC	Washington State Pollution Control Commission
WQA	water quality assessment
WQC	water quality criteria
WQS	water quality standards
WRCC	Western Regional Climate Center
WRIA	Water Resource Inventory Area
WSDOH	Washington State Department of Health
WSDOT	Washington State Department of Transportation
ww	wet weight
WWTP	wastewater treatment plant

Executive Summary

This document presents the results of the remedial investigation (RI) for the Lower Duwamish Waterway (LDW), located in Seattle, Washington. The RI provides information on the extent of contamination and the risks to humans and the environment posed by the contamination. The RI will be used to help make decisions about necessary sediment cleanup actions and other actions to manage risks.

In 2000, the City of Seattle, King County, the Port of Seattle, and The Boeing Company, working collectively as the Lower Duwamish Waterway Group, agreed in an Administrative Order on Consent to conduct a remedial investigation/feasibility study (RI/FS) for the LDW with oversight by the US Environmental Protection Agency (EPA) and the Washington State Department of Ecology (Ecology). In September 2001, the LDW was formally listed as a Superfund site; and in February 2002, the LDW was formally listed as a Washington Model Toxics Control Act (MTCA) site.

This RI presents the results of many years of investigations conducted for the LDW study area, which extends from the southern tip of Harbor Island to just south of the Upper Turning Basin (Map ES-1). The RI data will support the analyses to be conducted in the FS to allow EPA and Ecology to make cleanup decisions for the LDW.

This RI describes what is known about the LDW, including:

- The history of the LDW as well as the environmental setting, habitat, and uses
- The deposition and transport of sediment within the LDW
- The distribution of contamination in the LDW, including concentrations of chemicals in sediment, water, and tissues
- Information regarding potential historical and ongoing sources of chemicals to the LDW as well as the source control and identification strategy
- The results of the baseline human health risk assessment (HHRA) and ecological risk assessment (ERA), which assess risks to people and ecological species from contamination within the LDW prior to remedial actions

The specific topics addressed in each section of the RI are summarized below. Details are provided in the RI, its appendices, and attachments.

ES.1 ENVIRONMENTAL SETTING

The LDW is an estuary that has been extensively modified over the past 100 years. Diversion of two major rivers (i.e., White River in 1906 and Black River in 1916) approximately 100 years ago reduced the flow volume and watershed area by about 70%. Today, the Green River is the primary source of water for the Duwamish River.

Many of the natural curves of the estuary were eliminated when the navigation channel was created in the early 20th century. Prior to 1920, the King County Commercial Waterway District No. 1 had dredged the LDW to an Upper Turning Basin at River Mile (RM) 4.6. This Upper Turning Basin serves as a trap for most sediment being carried downstream by the Green/Duwamish River. Frequent dredging is still necessary to remove sediment from this area and thereby keep it from being transported into the lower reaches of the LDW. The construction of the Howard Hansen Dam, approximately 65 mi upstream of the LDW, in 1961 effectively decreased peak flows in the river system, which now rarely exceed 12,000 cubic feet per second.

Today, the LDW is maintained as a federal navigation channel by the US Army Corps of Engineers. The shoreline along the majority of the LDW has been developed for industrial and commercial operations. Common shoreline features include constructed bulkheads, piers, wharves, sheet piling walls, buildings that extend over the water, and steeply sloped banks armored with riprap or other fill material. Despite the highly modified river habitat, the LDW is home to diverse communities of fish, birds, mammals, and invertebrate species.

The LDW corridor is one of the City of Seattle's primary industrial areas. Current land use, zoning requirements, and land ownership within most of this corridor are consistent with the characteristics of an active industrial waterway. Two neighborhoods, South Park and Georgetown, are located to the west and east, respectively, of the LDW. These neighborhoods support a mixture of residential, recreational, commercial, and industrial uses. The LDW supports considerable commercial navigation and is also used for various recreational activities such as boating, kayaking, fishing, and beach play. Several public parks and publicly accessible shoreline areas exist. There are plans to create additional recreational and habitat opportunities in the LDW corridor. The LDW is also one of the locations of the Muckleshoot Tribe's commercial, ceremonial, and subsistence fishery for salmon. The Suquamish Tribe actively manages aquatic resources north of the Spokane Street Bridge, located just north of the LDW. The Duwamish Tribe uses Herring's House Park and other parks along the Duwamish for cultural gatherings.

Intertidal habitats are dispersed in relatively small patches downstream of RM 3.0, with the exception of Kellogg Island, which represents the largest contiguous area of intertidal habitat remaining in the LDW. There is great interest in restoring and improving the intertidal habitat in the LDW. Several restoration projects have already been completed, some of which included the creation of public parks.

ES.2 SEDIMENT DYNAMICS

Several studies of LDW sediment movement have been conducted since the 1970s, but the most extensive work has been done in the past 3 years as part of the RI. This type of analysis will be used in the FS to estimate the potential for contaminated surface sediment to be buried by sediment from upstream in the future and to estimate areas and depths of erosion. This work included the collection of new data to be used in conjunction with historical data to develop a sediment transport model (STM) for the LDW. This model, as well as other lines of evidence, was used to develop a physical conceptual site model (CSM) for the LDW. The key components of the CSM are:

- On annual time scales, more sediment is added (deposited) throughout the LDW than is removed (eroded).
- The rate at which sediment is deposited is generally higher in deeper areas, especially the navigation channel, than in shallower areas, such as intertidal benches.
- High-flow events that occur periodically (e.g., one event every 2 years or less frequently), generally as a result of high rainfall events, have the greatest influence on sediment erosion. Net erosion (when the amount of erosion exceeds that of deposition) occurs at 20% or less of the LDW during such events. During such high-flow events, approximately 6% of the LDW experiences net erosion of 10 cm or greater (maximum depth of 21 cm).[1] Over much of the LDW, erosion depths are much less, particularly between RM 0.0 and RM 2.0, where net erosion is predicted to be negligible.
- Tugboats in the navigation channel tend to mix a thin layer of sediment (i.e., 1 to 2 cm) repeatedly over certain areas where these boats travel. Tugboats have the potential to erode sediment to a greater depth while maneuvering over relatively small areas outside the navigation channel (e.g., docks and berthing areas).
- The STM developed for the RI/FS (QEA 2008) is an important tool for predicting the changes, both erosion and deposition, that occur on the sediment bed. Simulations were run for both short (days or weeks) and long (years) time periods. The STM was also used to estimate changes in the physical composition of the sediment surface layer over time as sediment from upstream and lateral sources (e.g., storm drains) becomes incorporated into the sediment bed. Based on the results of the modeling conducted to date, approximately

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[1] These depths are similar to the 10-cm (3.9-in.) depth used to characterize surface sediment within the LDW and the 10-cm depth of the biologically active zone, which was based in part on the sediment profile imaging data (Ecology 2007n).

99% of the total external sediment particle load[2] to the LDW comes from the Green River, upstream of the LDW.

ES.3 NATURE AND EXTENT OF CONTAMINATION

The presence of chemical contamination in the LDW has been known for many years, prompting numerous environmental studies. From 1990 to 2001, approximately 1,200 surface sediment samples, 230 subsurface sediment samples, 90 fish and shellfish tissue samples, and 480 surface water samples, were collected and analyzed for metals and organic compounds. A Phase 1 RI (Windward 2003a) was initially produced based solely on information available at that time. Additional chemical data needs were subsequently identified (Windward 2003f), and a plan for filling those data needs was prepared (Windward 2004h) and approved by EPA and Ecology. Since 2003, approximately 900 additional samples were collected and analyzed as part of the RI to characterize chemical contamination in the following media:

- Juvenile chinook salmon tissue
- Fish, clam, and crab tissue
- Benthic invertebrate tissue
- Seep water (tidally fluctuating groundwater seeping from the banks along the river)
- Surface sediment (top 10 cm or 3.9 in.)
- Subsurface sediment (below the top 10 cm)
- Porewater (water in spaces between sediment particles)

In addition, samples of surface water[3] and source-tracing samples (e.g., sediments from catch basins and in-line sediment traps from stormwater and combined sewer overflow basins) have also been collected and analyzed.

Based on surface sediment data, the LDW can be characterized as having localized areas with relatively high chemical concentrations (hot spots) separated by relatively large areas with lower chemical concentrations (Table ES-1). Many of the highest concentrations of key chemicals are in areas that were identified as candidates for early action (known as early action areas [EAAs]) (Maps ES-2a, ES-2b, and ES-2c)

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[2] Particle loading should be distinguished from chemical loading. Chemical loads from lateral sources could have a significant influence on the sediment and water quality in the vicinity of a lateral source, even though the relative magnitude of particle loading from those sources is very small compared with particle loading from upstream sources.

[3] Surface water samples were collected by King County for analysis of PCB congeners in 2005. Samples representing potential sources to the LDW are discussed in Section 9 (samples from combined sewer overflows in Section 9.4.4.2; sediment samples from the storm drain and combined sewer overflow basins in Section 9.4.4.6; and groundwater samples in Section 9.4.6.1).

(Windward 2003e). Polychlorinated biphenyls (PCBs), various metals, polycyclic aromatic hydrocarbons (PAHs), and phthalates were frequently detected in surface sediments (Maps ES-3a, ES-3b, and ES-3c). Samples from a smaller number of locations in the LDW were analyzed for dioxins and furans; at least one dioxin or furan congener was detected in each sample. Many other organic chemicals, such as volatile organic compounds (VOCs), other semivolatile organic compounds, and pesticides, were less frequently or rarely detected.

Table ES-1. Summary statistics for key chemicals in surface sediment

CHEMICAL	UNIT	DETECTION FREQUENCY	CONCENTRATION			
			MEAN	MEDIAN	95TH PERCENTILE	MAXIMUM
Total PCBs	µg/kg dw	1,243/1,327	1,170	137	4,300	220,000
Arsenic	mg/kg dw	794/852	17	11	30	1,100
cPAHs[a]	µg/kg dw	780/828	460	260	1,500	11,000
Dioxin and furan TEQ	ng/kg dw	54/54	82.1	10.4	490	2,100
BEHP	µg/kg dw	674/832	590	230	2,400	14,000

Note: summary statistics were calculated assuming one-half the reporting limit for non-detect results.

[a] cPAH concentrations are expressed in terms of benzo(a)pyrene equivalents.

dw – dry weight
BEHP – bis(2-ethylhexyl) phthalate
cPAH – carcinogenic polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

The distribution of chemicals within subsurface sediment has also been characterized. Chemicals frequently detected in subsurface sediment were similar to those detected in surface sediment (Table ES-2). In many areas, the highest chemical concentrations were buried under surface sediment with lower concentrations. These results are generally consistent with the STM results, which emphasize the importance of sediment transported from the Green River to the sediment dynamics within the LDW and the fact that concerted efforts to control sources have resulted in the concentrations of chemicals in current releases generally being lower than those in historical releases. There are many exceptions to this general trend, which could be the result of localized chemical sources or lower than predicted net sedimentation rates because of site-specific conditions the STM did not evaluate.

Table ES-2. Summary statistics for key chemicals in sediment cores

CHEMICAL	UNIT	DETECTION FREQUENCY[a]	CONCENTRATION				DEPTH INTERVAL OF MAXIMUM CONCENTRATION (ft)[b]
			MEAN	MEDIAN	95TH PERCENTILE	MAXIMUM	
Total PCBs	µg/kg dw	609/821	3,000	170	5,600	890,000	0.3 – 1.5
Arsenic	mg/kg dw	267/325	40	12	63	2,000	2 – 4
cPAHs[c]	µg/kg dw	252/304	400	190	1,500	7,000	1 – 2
Dioxin and furan TEQ	ng/kg dw	26/26	27.2	14.4	170	194.0 J	4 – 6
BEHP	µg/kg dw	216/306	500	230	1,800	5,100	0 – 3

Note: summary statistics were calculated assuming one-half the reported or calculated non-detect results.

[a] Total number of samples represents all samples collected from any subsurface interval at all locations. Statistics are calculated based on all samples.

[b] Depth interval with highest concentration for a given chemical within any single core within the LDW.

[c] cPAH concentrations are expressed in terms of benzo(a)pyrene equivalents.

dw – dry weight
BEHP – bis(2-ethylhexyl) phthalate
cPAH – carcinogenic polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Some areas have high chemical concentrations in both subsurface and surface sediment. This distribution of contamination generally is found in areas estimated to have low net sedimentation rates by the STM. In a few areas where higher sedimentation was predicted, the presence of high chemical concentrations near the surface could be the result of localized disturbance events or recent or ongoing sources of contamination.

Tissue samples of many different fish and invertebrate species have been collected and analyzed. Most of the tissue data represent English sole, shiner surfperch, Pacific staghorn sculpin, juvenile chinook salmon, Dungeness and slender crabs, soft-shell clams, and small invertebrates that live in the sediment, such as amphipods and marine worms. These species were selected because they were assumed to be representative of species that could be consumed by people, fish, or wildlife within the LDW. Their tissues were analyzed for a wide variety of chemicals.

Mean total PCB concentrations were highest for Dungeness crab hepatopancreas and English sole whole-body tissue, followed by shiner surfperch (Table ES-3). Mean total PCB concentrations were lowest for mussels. Inorganic arsenic concentrations in softshell clams were much higher than in fish and crabs collected in the LDW and in clams from Puget Sound. Carcinogenic PAH (cPAH) concentrations were highest in clams, mussels, and benthic invertebrates, and phthalates were frequently detected in clams and benthic invertebrates. Most other organic chemicals were infrequently detected.

Table ES-3. Summary statistics for key chemicals in tissue

Chemical and Tissue Type	Unit	Detection Frequency[a]	Concentration		
			Minimum	Mean	Maximum
Total PCBs					
Juvenile chinook salmon (whole body)	µg/kg ww	24/24	6.9	140	1,200
English sole (whole body)	µg/kg ww	67/67[b]	300	1,700	4,700
English sole (fillet with skin)	µg/kg ww	26/26	170	860	2,010
English sole (fillet without skin)	µg/kg ww	15/15	79	230	530
Shiner surfperch (whole body)	µg/kg ww	78/78	200 J	1,300	18,400 J
Pacific staghorn sculpin (whole body)	µg/kg ww	28/28	430	900	2,800
Starry flounder (whole body)	µg/kg ww	6/6	156	380	660
Starry flounder (fillet with skin)	µg/kg ww	2/2	63	260	450
Striped perch (fillet with skin)	µg/kg ww	3/3	164 J	320 J	630 J
Pile perch (fillet with skin)	µg/kg ww	1/1	300	300	300
Dungeness crab (edible meat)	µg/kg ww	14/17	15	130	300
Dungeness crab (hepatopancreas)	µg/kg ww	11/11	280	2,000	5,500
Dungeness crab (whole body)[c]	µg/kg ww	16/16	97 M	890	1,900 JM
Red rock crab (edible meat)	µg/kg ww	2/2	85 J	120	164 J
Red rock/Dungeness crab (edible meat)	µg/kg ww	1/1	60 J	60	60 J
Slender crab (edible meat)	µg/kg ww	19/19	27	150	390
Slender crab (hepatopancreas)	µg/kg ww	11/11	250	940	2,190 J
Slender crab (whole body)[c]	µg/kg ww	19/19	100 M	510	838 JM
Clams (non-depurated)	µg/kg ww	20/20	15 J	130	580 J
Clams (depurated)	µg/kg ww	6/6	14 J	98	270
Amphipods	µg/kg ww	4/4	106	230	410
Mussels (wild)	µg/kg ww	18/22	16	34	60
Mussels (transplanted)	µg/kg ww	13/32	25.9	27	73.1
Benthic invertebrates	µg/kg ww	19/20	60 J	270	1,400
Inorganic Arsenic					
English sole (whole body)	mg/kg ww	7/7	0.020	0.051	0.090
English sole (fillet with skin)	mg/kg ww	6/7	0.003	0.004	0.006

Chemical and Tissue Type	Unit	Detection Frequency[a]	Concentration		
			Minimum	Mean	Maximum
Shiner surfperch (whole body)	mg/kg ww	8/8	0.020	0.070	0.16
Starry flounder (whole body)	mg/kg ww	1/1	0.09	0.09	0.09
Starry flounder (fillet with skin)	mg/kg ww	1/1	nd	0.0015 U	nd
Striped perch (fillet with skin)	mg/kg ww	0/1	nd	0.005 U	nd
Pile perch (fillet with skin)	mg/kg ww	0/1	nd	0.005 U	nd
Dungeness crab (edible meat)	mg/kg ww	2/2	0.010 J	0.010	0.010
Dungeness crab (hepatopancreas)	mg/kg ww	2/2	0.050	0.070	0.090
Dungeness crab (whole body)[c]	µg/kg ww	2/2	0.022 JM	0.029	0.035 M
Slender crab (edible meat)	mg/kg ww	4/4	0.030	0.030	0.030
Slender crab (hepatopancreas)	mg/kg ww	4/4	0.080	0.24	0.33
Slender crab (whole body)[c]	µg/kg ww	4/4	0.046 M	0.098	0.123 M
Clams (non-depurated)	mg/kg ww	23/23	0.132	2.72	11.3
Clams (depurated)	mg/kg ww	15/15	0.720	3.37	9.300
cPAHs[e]					
English sole (whole body)	µg/kg ww	18/21	0.45 J	1.6	2.8
English sole (fillet with skin)	µg/kg ww	4/7	0.37 J	0.35[d]	0.53
English sole (fillet without skin)	µg/kg ww	0/6	nd	8.9 U	nd
Shiner surfperch (whole body)	µg/kg ww	24/27	0.37 J	3.1[d]	2.2
Pacific staghorn sculpin (whole body)	µg/kg ww	1/24	36 J	130[d]	36 J
Starry flounder (whole body)	µg/kg ww	3/3	0.47 J	0.59 J	0.66 J
Starry flounder (fillet with skin)	µg/kg ww	1/1	0.64 J	0.64 J	0.64 J
Striped perch (fillet with skin)	µg/kg ww	1/1	0.43 J	0.43 J	0.43 J
Pile perch (fillet with skin)	µg/kg ww	1/1	0.43 J	0.43 J	0.43 J
Dungeness crab (edible meat)	µg/kg ww	6/9	0.54 J	3.7 J	0.84 J
Dungeness crab (hepatopancreas)	µg/kg ww	2/4	0.73 J	6.5[d]	0.73 J
Dungeness crab (whole body)[c]	µg/kg ww	7/9	0.60 JM	2.6[d]	2.4 JM
Slender crab (edible meat)	µg/kg ww	2/12	0.33	0.35	0.63 J
Slender crab (hepatopancreas)	µg/kg ww	4/4	0.68 J	0.70	0.71

CHEMICAL AND TISSUE TYPE	UNIT	DETECTION FREQUENCY[a]	CONCENTRATION MINIMUM	MEAN	MAXIMUM
Slender crab (whole body)[c]	µg/kg ww	12/12	0.45 M	0.65	0.67 M
Clams	µg/kg ww	14/14	6.8	15	44
Amphipods	µg/kg ww	0/4	nd	22 U	nd
Benthic invertebrates	µg/kg ww	20/20	4.2 J	44	290 J
Mussels (wild)	µg/kg ww	11/22	30	23[d]	33
Mussels (transplanted)	µg/kg ww	16/34	28.9	24[d]	34.6
BEHP					
English sole (whole body)	µg/kg ww	1/27	170	600[d]	170
English sole (fillet with skin)	µg/kg ww	2/7	1,100	370[d]	1,300 J
English sole (fillet without skin)	µg/kg ww	0/6	nd	4.9 U	nd
Shiner surfperch (whole body)	µg/kg ww	5/34	280 J	650 J	2,100 J
Pacific staghorn sculpin (whole body)	µg/kg ww	0/24	nd	1,700 U	nd
Starry flounder (whole body)	µg/kg ww	0/3	nd	33 U	nd
Starry flounder (fillet with skin)	µg/kg ww	0/1	nd	34 U	nd
Striped perch (fillet with skin)	µg/kg ww	0/1	nd	34 U	nd
Pile perch (fillet with skin)	µg/kg ww	0/1	nd	34 U	nd
Dungeness crab (edible meat)	µg/kg ww	0/9	nd	31 U	nd
Dungeness crab (hepatopancreas)	µg/kg ww	0/4	nd	36 U	nd
Dungeness crab (whole body)[c]	µg/kg ww	0/9	nd	15.6 U	nd
Slender crab (edible meat)	µg/kg ww	0/12	nd	60 U	nd
Slender crab (hepatopancreas)	µg/kg ww	1/4	100 J	70[d]	100 J
Slender crab (whole body)[c]	µg/kg ww	3/12	80 JM	45[d]	80 JM
Benthic invertebrates	µg/kg ww	5/20	1,100 J	1,400	2,200 J
Amphipods	µg/kg ww	2/4	170	180	530
Clams	µg/kg ww	10/14	56 J	140	220 J
Mussels (wild)	µg/kg ww	2/22	28	17[d]	190
Mussels (transplanted)	µg/kg ww	0/34	nd	8.6 U	nd

Note: Mean concentrations were calculated assuming one-half the reported or calculated non-detect results.

[a] This table includes all tissue data in the RI dataset (collected between 1995 and 2007; see Table 4-8).

[b] Number of samples includes 32 whole-body samples and 10 calculated “whole-body” samples. Concentrations in “whole-body” samples were estimated using results from separate analyses of fillet and remainder composite samples (i.e., all remaining tissue and fluids after fillets were removed from the specimens). The estimated English sole “whole-body” concentrations were based on the relative weights and total PCB concentrations in skin-on fillet and remainder tissues collected in 2005.

[c] Data from hepatopancreas composite samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Whole-body (i.e., edible meat plus hepatopancreas) crab concentrations were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weight of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data).

[d] The mean concentration is generated with both detect and non-detect data. Non-detect data were included at half RL.

[e] cPAH concentrations are expressed in terms of benzo(a)pyrene equivalents.

BEHP – bis(2-ethylhexyl) phthalate
cPAH – carcinogenic polycyclic aromatic hydrocarbon
J – estimated concentration
M – calculated value
na – not applicable
nd – not detected
PCB – polychlorinated biphenyl
U – not detected
ww – wet weight

Tissue samples collected in 2004 were analyzed for PCB Aroclors by Columbia Analytical Services, Inc.; samples collected in 2005, 2006, and 2007 were analyzed by Analytical Resources, Inc. The PCB mixtures in samples collected from 2005 to 2007 appeared to have had a higher proportion of the more highly chlorinated PCB congeners than did those collected in 2004.

Aroclor wet-weight and lipid-normalized tissue data as a whole suggested that total PCB concentrations in 2004 were higher than those in more recent years; total PCB concentrations appear to be decreasing based on Aroclor data. Trends based on the limited PCB congener data are more uncertain. PCB congener data suggest that total PCB concentrations (wet weight) in English sole collected from Area T1 (RM 0.2 to RM 1.0) and possibly from Area T2 (RM 1.6 to RM 2.4) may be decreasing, although the sample sizes are small. Total PCB concentrations (PCB congener sums, wet weight) in crabs collected from Areas T1 and T3 (RM 2.9 to RM 3.7) were also consistently lower in 2007 than 2004. The wet-weight PCB congener data suggest that temporal trends in total PCB concentrations in shiner surfperch are uncertain. The lipid-normalized PCB congener data suggest potential downward trends in Area T3 for English sole, in Area T2 for shiner surfperch, and in Areas T1 and T3 for Dungeness crab. Trends in PCB congener data could not be evaluated for Area T4 (RM 4.2 to RM 5.0) because PCB congener analyses were conducted on tissue samples from that area only in 2004.

Relationships between total PCB concentrations in tissue and sediment were investigated using regression analyses. Co-located tissue and sediment samples collected in 2004 were used for the clam and benthic invertebrate analyses, and surface sediment spatially weighted average concentrations (SWACs) and tissue data collected in 2004 and 2005 were used for the fish and crab analyses. Total PCB concentrations (wet weight) in clams and benthic invertebrates were significantly correlated with total PCB concentrations (dry weight) in sediment. English sole and crabs, which were composited from larger sampling areas than those of Pacific staghorn sculpin and shiner surfperch, appear to have somewhat larger foraging ranges based on several lines of evidence. Concentrations of total PCBs in English sole and crab tissues were not correlated with total PCB concentrations in sediment on a sampling area scale (on either a wet-weight or lipid-normalized basis). Both wet-weight and lipid-normalized total PCB concentrations in Pacific staghorn sculpin tissues were correlated with total PCB concentrations in sediment on a subarea scale (0.3 RM) and on an area scale (approximately 1 RM). Both wet-weight and lipid-normalized total PCB concentrations in shiner surfperch tissue were correlated with total PCB concentrations in sediment on a subarea scale but not on an area scale.

The relationships between arsenic concentrations in sediment and tissue were statistically evaluated only for benthic invertebrates and clams. For benthic invertebrates, the relationship was significant, but there was a high degree of uncertainty in the relationship for clams. Mean total arsenic concentrations in English sole and Pacific staghorn sculpin were similar among areas but were slightly higher in Area T1. Total arsenic concentrations in shiner surfperch tissue were similar among all areas. In crab tissue, the highest total arsenic concentration was in Area T4, but only one Dungeness crab sample was collected in this area. When this sample is excluded, the highest mean total arsenic concentrations in crabs were in samples from Area T1.

The relationships between cPAH concentrations in sediment and tissue were evaluated only for clams; no strong relationship was found between cPAH concentrations in clam tissue and co-located sediment. cPAH concentrations in whole-body English sole samples were much lower in Area T4 than in Areas T1 and T2. For shiner surfperch, mean concentrations were lower in Area T4 than in Area T3. In Dungeness crab samples, cPAH concentrations were similar among all areas, except for the single sample collected from Area T4, which had a higher concentration.

Relationships between bis(2-ethylhexyl) phthalate (BEHP) concentrations in sediment and tissue were not evaluated because BEHP was infrequently detected in tissue. Relationships between dioxin and furan concentrations in sediment and tissue were not evaluated because dioxins and furans were not analyzed in tissue samples from the LDW.

Chemical concentrations in surface water, seep water, and porewater were also summarized in the RI. A large number of surface water grab samples were collected by King County in 1996 and 1997 and analyzed for metals and SVOCs. More recently (2005), water samples were collected and analyzed for PCB congeners using methods that are more sensitive than those used previously. Total PCB concentrations in whole-water samples ranged from 0.13 to 3.2 ng/L, with the lowest concentrations detected during periods when the flows were highest.

Seep water and porewater data were collected in areas along the banks of the LDW where EPA and Ecology considered the likelihood of contamination to be the greatest. These data were collected instead of additional groundwater data because they are more directly relevant to animals using the LDW. The results of seep water sampling showed that arsenic, copper, and zinc were detected at concentrations greater than the Washington State acute marine water quality criteria in seep samples from at least one location in the LDW. VOCs were detected in both seep water and porewater in a localized area near one facility with well-documented VOC contamination in groundwater (i.e., Great Western International). VOCs were also detected in some of the porewater samples collected near the boundary between Boeing Plant 2 and Jorgensen Forge.

ES.4 BASELINE ECOLOGICAL RISK ASSESSMENT

The baseline ERA presents risk estimates for benthic invertebrate, crabs, fish, and wildlife species that may be exposed to chemicals in the LDW. Chemicals of potential concern (COPCs) were first identified through a conservative risk-based screening process so that risk analyses could focus on chemicals more likely to be of concern.

There were 10 receptors of concern (ROCs)[4] evaluated in the ERA (i.e., the benthic invertebrate community, crabs, juvenile chinook salmon, Pacific staghorn sculpin, English sole, spotted sandpiper, great blue heron, osprey, river otter, and harbor seal). ROCs are species selected to represent larger groups of animals because not all species can be individually evaluated in the ERA. The risk estimates were based either on COPC concentrations in sediment, water, and aquatic biota from the LDW or on estimated dietary chemical doses, depending on the species and chemical.

For crabs, fish, and wildlife, the site-related chemical concentrations or doses were compared with concentrations or doses from the scientific literature that have been shown to cause specific harmful effects in the same or related species. The effects of primary concern are those that decrease survival, growth, or reproduction.

To evaluate risks to the benthic macroinvertebrate community, chemical concentrations in sediment were compared with the Washington State Sediment Management Standards (SMS), which provides for both chemical and biological standards. The goal of the SMS is to reduce and ultimately eliminate adverse effects on biological resources (Washington Administrative Code [WAC] 173-204-100). For benthic invertebrates living in intertidal and subtidal sediments, sediment chemistry and site-specific toxicity test results indicated that no adverse effects are expected in approximately 75% of the LDW area (345 ac), based on chemical concentrations in surface sediment that were less than the sediment quality standards (SQS) of the SMS and toxicity test results (Maps ES-3a, ES-3b, ES-3c, and ES-4). There is a higher likelihood for adverse effects in approximately 7% of the LDW area (34 ac), which was found to have chemical concentrations or biological effects in excess of the cleanup screening levels (CSLs) of the SMS. The remaining 18% of the LDW area (82 ac) had chemical concentrations or biological effects between the SQS and CSL, indicating that

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[4] Key considerations for selecting ROCs include the potential for direct or indirect (e.g., ingestion of fish or invertebrates) exposure to sediment-associated chemicals, human and ecological significance, site usage, sensitivity to COPCs at the site, susceptibility to biomagnification of COPCs (i.e., higher-trophic-level species), and data availability.

risks to benthic invertebrate communities are less certain in these areas than in areas with concentrations greater than one or more CSL values.

Forty-one chemicals exceeded the SQS in one or more surface sediment samples. Concentrations of total PCBs, BEHP, and butyl benzyl phthalate exceeded SMS chemical criteria more frequently than any other chemicals. No other chemicals exceeded the SQS in more than 5% of the surface sediment samples, although the ratio of the maximum detected concentration to the SQS was greater than 10 for some chemicals (e.g., lead, acenaphthene, and 1,4-dichlorobenzene).

Of the 48 sediment samples tested for toxicity during the RI, 11 exceeded the SQS biological effects criteria, 19 exceeded the CSL biological effects criteria, and 18 did not exceed either the SQS or CSL biological effects criteria.

VOCs were analyzed in sediment porewater at two locations in areas where VOCs were known to be present at high concentrations in groundwater on upland properties adjacent to the LDW. One VOC (cis-1,2-dichloroethene) was detected at a concentration higher than the no-observed-effect concentration for marine invertebrates in porewater samples collected from high intertidal areas at one of the two locations (Great Western International at RM 2.4); the maximum concentration was lower than the lowest-observed-effect concentration. Thus, it is uncertain whether the cis-1,2-dichloroethene concentration is sufficiently high within this area of the LDW to result in adverse effects. Because this area was considered to be a worst-case exposure area with respect to the potential for adverse effects on benthic invertebrates from VOCs, and because other areas with groundwater data had lower VOC concentrations, it was concluded that there was a low likelihood of appreciable risks from VOCs in sediment porewater elsewhere in the LDW.

Risks to benthic invertebrates from tributyltin (TBT) were considered to be low based on an evaluation of imposex in field-collected gastropods, as well as a comparison of TBT concentrations in benthic invertebrate tissue samples to tissue effect concentrations from the scientific literature. There was a significant regression relationship between TBT concentrations in benthic invertebrates and co-located surface sediment samples. This regression was used to estimate an LDW-wide exposure point concentration (EPC) of TBT in benthic invertebrate tissue for comparison to the effect concentrations from the literature. TBT concentrations in benthic invertebrate tissue were less than no-observed-effect concentrations in all samples analyzed and as estimated using the regression.

Total PCB concentrations in benthic invertebrate tissue were significantly correlated with total PCB concentrations in co-located sediment. Total PCB concentrations in benthic invertebrate tissues were compared with available toxicity data in the uncertainty assessment; use of this alternative method for risk evaluation did not identify additional locations with the potential for adverse effects resulting from PCBs.

The relationship between metal concentrations in benthic invertebrates and co-located sediment was evaluated and the only significant regression relationship was for arsenic. This relationship was used to estimate an LDW-wide EPC of arsenic in benthic invertebrate tissue for use in evaluating risk to fish from ingestion of benthic invertebrates.

For the other receptors evaluated as ROCs, hazard quotients (HQs) were calculated based on a range of effects data, including no-observed-adverse-effect levels (NOAELs) and lowest-observed-adverse-effect levels (LOAELs). The highest LOAEL-based HQ for fish, crabs, and wildlife species was 5.5 (for lead in spotted sandpiper) (Table ES-4).

Table ES-4. Summary of ecological risks for fish and wildlife species

Chemical	Receptor of Concern	NOAEL-Based HQ	LOAEL-Based HQ
COCs with LOAEL-Based HQs ≥ 1.0[a]			
Total PCBs	crab	**10**	**1.0**
	English sole	**4.9 – 25**	0.98 – **5.0**
	Pacific staghorn sculpin	**1.5 – 19**	0.30 – **3.8**
	river otter	**5.8**	**2.9**
PCB TEQs	spotted sandpiper	**1.9 – 15**	0.18 – **1.5**
Cadmium	juvenile chinook salmon	**5.0**	**1.0**
	English sole	**6.1**	**1.2**
	Pacific staghorn sculpin	**3.0 – 5.2**	0.60 – **1.0**
Chromium	spotted sandpiper	**1.3 – 8.8**	0.26 – **1.8**
Copper	spotted sandpiper	0.62 – **1.5**	0.45 – **1.1**
Lead	spotted sandpiper	0.58 – **19**	0.17 – **5.5**
Mercury	spotted sandpiper	**1.1 – 5.3**	0.21 – **1.0**
Vanadium	English sole	**5.9**	**1.2**
	Pacific staghorn sculpin	**3.2 – 5.9**	0.65 – **1.2**
	spotted sandpiper	**2.0 – 2.7**	**1.0 – 1.4**
COPCs with NOAEL-Based HQs ≥ 1.0 and LOAEL-Based HQs < 1.0[b]			
Total PCBs	spotted sandpiper	0.51 – **2.0**	0.18 – 0.71
PCB TEQs	osprey	**1.6**	0.16
	river otter	**4.5**	0.59
Arsenic	juvenile chinook salmon	**1.1**	0.73
	English sole	**1.2**	0.80
	crab	**3.9**	na
Benzoic acid	English sole	**1.5**	na
	Pacific staghorn sculpin	**2.1**	na
Cadmium	Pacific staghorn sculpin	**3.0 – 4.9**	0.60 – 0.98
Chromium	juvenile chinook salmon	**2.1**	na
	English sole	**1.1**	na

CHEMICAL	RECEPTOR OF CONCERN	NOAEL-BASED HQ	LOAEL-BASED HQ
Copper	Juvenile chinook salmon	**1.9**	0.93
	English sole	**1.9**	0.93
	Pacific staghorn sculpin	0.9 – **1.5**	0.45 – 0.77
Mercury	river otter	**2.8**	0.57
TBT	Pacific staghorn sculpin	**1.6** – **2.9**	0.18 – 0.33
Vanadium	juvenile chinook salmon	**4.0**	0.79
Zinc	crab	**2.5**	0.91

Note: HQs for fish are the highest HQs in cases where more than one approach was used. **Bold** identifies NOAEL-based HQs greater than 1.0 or LOAEL-based HQs greater than or equal to 1.0.

[a] The LOAEL-based HQs for endrin were 1.2 and 3.1 for English sole and Pacific staghorn sculpin, respectively, based on risk calculations discussed in the uncertainty section. These calculations were presented only in the uncertainty section because of analytical interferences from PCB Aroclors in the pesticide analyses of LDW tissue samples, resulting in uncertainties in pesticide identification and a high bias in pesticide concentrations.

[b] The NOAEL-based HQs were ≥1 for the following COPC/ROC pairs based on risk calculations discussed in the uncertainty section: 1) total DDTs and spotted sandpiper (2.6 to 4.3), 2) endrin and juvenile chinook salmon (3.6), 3) alpha-endosulfan and English sole (6.8) and Pacific staghorn sculpin (2.3), 4) beta-endosulfan and English sole (29) and Pacific staghorn sculpin (6.6), 5) endrin and juvenile chinook salmon (3.6), and 6) methoxychlor and crab (3.6). These calculations were presented in the uncertainty section because of analytical interferences from PCB Aroclors in the pesticide analyses, resulting in uncertainties in pesticide identification and a high bias in pesticide concentrations.

COPC – chemical of potential concern
DDT -dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TEQ – toxic equivalent

Chemicals of concern (COCs) were defined as chemicals with LOAEL-based HQs greater than or equal to 1, which indicates a potential for adverse effects. Seven chemicals were identified as COCs (PCBs, cadmium, chromium, copper, lead, mercury, and vanadium) for at least one fish and wildlife species (Table ES-4).

No quantitative ecological risk estimates were calculated for dioxins and furans; tissue samples were not analyzed for dioxins and furans because of the difficulties associated with assessing risks from dioxins and furans, the need for a large background dataset with which to compare site-specific data, and the paucity of background tissue data in the Puget Sound area. Remedial decisions to address dioxin and furan contamination in sediment will be based on MTCA and CERCLA regulations, including those specifically related to background (Ecology 2001a; EPA 2002d).

Based on the risk estimates, uncertainties discussed in the ERA, preliminary natural background concentrations, and COC concentrations in EAAs, chemicals were identified as risk drivers for ecological receptors in accordance with EPA (1998) and Ecology (WAC 173-340-703) guidance. The risk drivers from both the ERA and the HHRA will be the focus of remedial analyses in the FS. In consultation with EPA and Ecology, PCBs were identified as a risk driver for river otter because estimated exposure concentrations for river otter were greater than the LOAEL by a factor of 2.9 and uncertainties in the risk estimate were relatively low. In addition, 41 chemicals were selected as risk drivers for the benthic invertebrate community because concentrations of these 41 chemicals exceeded the SQS at one or more locations. COCs not selected as risk drivers will be addressed through focused evaluation in the FS; these chemicals may also be considered in remedial design for specific areas in or near the LDW and in the post-remedial monitoring program that is part of the 5-year review that EPA conducts at all Superfund sites where hazardous substances, pollutants, or contaminants remain in place after cleanup is completed.

ES.5 BASELINE HUMAN HEALTH RISK ASSESSMENT

The baseline HHRA estimated the risks people could face from exposure to chemicals in LDW sediments and seafood (exposure pathways). Direct contact with sediments during commercial netfishing, clamming, or beach play in the LDW and consumption of seafood from the LDW were identified as primary exposure scenarios. Quantitative risk estimates for other exposure scenarios, such as swimming, were also included in the HHRA but were calculated in a previous risk assessment, which suggested that risks from those scenarios were insignificant.

Reasonable maximum exposure (RME) estimates were calculated for all exposure scenarios to avoid underestimating risks. Consequently, risk estimates may be overestimated for many individuals. This approach is consistent with EPA's policy of "RME," which uses high-end, but plausible, estimates of exposure for assessing risks.

There are limited data available on the amount of resident seafood organisms currently being harvested and consumed from the LDW. A creel survey conducted by King County in the mid-1990s indicated that far less seafood was being harvested in the LDW compared with Elliott Bay and the Spokane Street Bridge just north of the LDW site boundary. More recently, the Washington State Department of Health (WSDOH) has issued health advisories for resident fish and shellfish in the LDW, warning the public that any level of seafood consumption from the LDW is unsafe because of elevated total PCB concentrations in these species (WSDOH 2005). WSDOH (Office of Food Safety and Shellfish) and Public Health—Seattle and King County also advise against the consumption of bivalves (i.e., clams, mussels, and oysters) from all of King County, including the LDW, because of pollution concerns (WSDOH 2005). There are no recent data on seafood consumption rates specific to the LDW, but such data should not be used to estimate potential future risks because current consumption rates may be suppressed. There are several possible explanations for such suppression, including the current advisory against the consumption of resident fish and shellfish, media coverage of the published risks from seafood consumption in the LDW, and the close proximity of more desirable fishing locations outside the LDW. Therefore, EPA and Ecology required the application of seafood consumption rates that have been developed for adult and child consumers based on seafood harvest from King County or Puget Sound. The tribal seafood consumption rates represent relatively high rates that might occur for harvest from the LDW at some time in the future, and were identified by EPA as the most appropriate RME seafood consumption scenarios for tribal members, although they likely overestimate the current consumption of resident seafood organisms from the LDW. An additional RME scenario required by EPA and Ecology was based on seafood consumption by adult Asian and Pacific Islanders (APIs). There is considerable uncertainty about the applicability of some of the seafood consumption rates to this HHRA, as discussed in Section 6.4.

Although salmon are a highly preferred and consumed fish from the LDW and tissue data were available for salmon, human health risks were not calculated for the consumption of adult salmon. PCBs accumulated in juvenile salmon during the time they spend in the LDW constitute only a very small fraction of the PCBs in adult salmon. Most of the PCBs in adult salmon result from foraging activities in Puget Sound or the Pacific Ocean.

Other scenarios with exposure assumptions higher or lower than those of the RME scenarios were also evaluated to characterize the range of risk estimates. For seafood consumption, these other scenarios included consumption rates estimated for the Suquamish Tribe, "average exposure" scenarios using central tendency consumption rate estimates, and a "unit risk" scenario based on an assumed one seafood meal per month. RME risks are generally used to evaluate remedial actions at a site. Risks estimated using Suquamish Tribe consumption rates represent an upper bound on tribal seafood consumption risks. Risks estimated using average consumption rates are a measure of the most likely risks posed by chemicals in seafood. The one-meal-per-month exposure scenario is not meant to actually describe seafood consumption that is occurring on the LDW (there are limited data[5] on current recreational seafood consumption rates for the LDW). Instead, it is intended to allow individuals to evaluate their own exposure depending on how many meals of seafood of different types collected from the LDW are consumed per month. For example, if an individual consumed two meals per month, then the risk for this individual would be two times the one-meal-per-month risk estimate.

There are no LDW-specific data to estimate the degree to which humans may currently be directly exposed to sediments via beach play or clamming activities. The exposure scenarios assumed for these activities were intended to represent exposures for a health-protective estimation of risks. The tribal netfishing scenario, on the other hand, reflects exposure conditions that could occur under current tribal fishing activities within the LDW.

Once the exposure scenarios were selected, chemistry data for each COPC were compiled over the assumed exposure area (e.g., intertidal sediments for clamming). For the seafood consumption scenarios, data for several different species (e.g., English

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[5] In 1997, King County conducted a survey on seafood collection and consumption that included locations in the LDW.

sole, perch, crab, clams, mussels) were used to represent a typical seafood consumer's diet.

Based on health-protective exposure assumptions, estimated cancer risks in the LDW were determined to be highest for the seafood consumption scenarios (Table ES-5). The cumulative risk for all carcinogenic chemicals ranged from 7 in 10,000 (7×10^{-4}) to 3 in 1,000 (3×10^{-3}) for the RME seafood consumption scenarios (adult tribal RME based on Tulalip Tribes' data, child tribal RME based on Tulalip Tribes' data, and adult API RME), with the primary contributors being PCBs, arsenic, and carcinogenic PAHs (cPAHs). Seafood samples from the LDW were not analyzed for dioxins and furans, so risks from these chemicals are not included in seafood consumption risk estimates but were assumed to be unacceptable. The evaluation of non-cancer HQs indicates the potential for adverse effects other than cancer associated with seafood consumption, particularly for arsenic and PCBs (Table ES-5).

Table ES-5. Summary of risks associated with seafood ingestion scenarios

SCENARIO	INGESTION RATE (g/day)[a]	MEALS PER MONTH[b]	EXPOSURE DURATION (years)	EXCESS CANCER RISK	NON-CANCER HI[c]
Adult tribal RME (Tulalip data)	97.5	13.1	70	3×10^{-3}	47
Adult tribal CT (Tulalip data)	15	2.0	30	1×10^{-4}	5
Adult tribal (Suquamish data)	597.7	80	70	3×10^{-2}	348
Child tribal RME (Tulalip data)	39.0	5.2	6	7×10^{-4}	104
Child tribal CT (Tulalip data)	6.0	0.8	6	7×10^{-5}	10
Adult API RME	51.5	6.9	30	1×10^{-3}	35
Adult API CT	5.3	0.7	9	2×10^{-5}	2
One meal per month – benthic	7.5	1.0	30	1×10^{-4}	6
One meal per month – crab	7.5	1.0	30	4×10^{-5}	1
One meal per month – clam	7.5	1.0	30	2×10^{-4}	4
One meal per month – pelagic fish	7.5	1.0	30	2×10^{-4}	10

[a] Rates do not include consumption of adult salmon because these fish acquire most of their chemical body burden from outside the LDW.

[b] It is assumed that one meal is equal to 227 g (8 ounces).

[c] Total across all chemicals. This total is not directly interpretable for risk assessment because it includes hazard quotients across multiple endpoints. The values indicate that the HI may exceed 1 for individual endpoints.

API – Asian and Pacific Islander
CT – central tendency
HI – hazard index (a sum of the HQs for individual chemicals)
RME – reasonable maximum exposure

Cancer risks for direct sediment contact RME scenarios for netfishing, clamming, and beach play were much lower (Table ES-6). Risk estimates ranged from 5 in 1,000,000 (5×10^{-6}) to 1 in 10,000 (1×10^{-4}). COCs for these pathways included PCBs, arsenic, cPAHs, and dioxins and furans. None of the direct sediment contact RME scenarios had HQs greater than 1 for non-cancer effect, and therefore, non-cancer effects are not expected from direct contact with sediments.

Table ES-6. Summary of risks associated with direct sediment contact scenarios

SCENARIO	EXPOSURE AREA	INCIDENTAL SEDIMENT INGESTION RATE (g/day)	EXPOSURE FREQUENCY (days/yr)	EXPOSURE DURATION (years)	EXCESS CANCER RISK
Netfishing RME	all subtidal and intertidal	0.050	119	44	3×10^{-5}
Netfishing CT	all subtidal and intertidal	0.050	63	29	5×10^{-6}
Beach play RME	Area 1	0.20	65	6	2×10^{-5}
	Area 2				5×10^{-5}
	Area 3				3×10^{-5}
	Area 4				3×10^{-5}
	Area 5				8×10^{-6}
	Area 6				9×10^{-6}
	Area 7				5×10^{-6}
	Area 8				7×10^{-6}
Clamming 7 days per year	intertidal area accessible from the shore	0.1	7	30	1×10^{-6}
Tribal clamming RME scenario	intertidal area accessible from the shore or from a boat	0.1	120	64	1×10^{-4}
Tribal clamming 183 days per year	intertidal area accessible from the shore or from a boat	0.1	183	70	2×10^{-4}

Note: Non-cancer hazard quotients did not exceed 1 for any chemical and are therefore not shown in this table.
CT – central tendency
RME – reasonable maximum exposure

Nineteen chemicals were identified as COCs (a COC has a cancer risk estimate greater than 1 in 1,000,000 [1×10^{-6}] or an HQ greater than 1 for an RME scenario) for at least one RME seafood consumption scenario.[6] Five chemicals were also identified as COCs based on a cancer risk estimate greater than 1 in 1,000,000 (1×10^{-6}) for at least one direct sediment contact RME scenario.[7] Four of these COCs were identified as risk

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[6] The COCs based on seafood consumption scenarios were PCBs, arsenic, cPAHs, dioxins and furans, aldrin, BEHP, alpha-benzene hexachloride (BHC), beta-BHC, carbazole, total chlordane, total DDTs, dieldrin, gamma-BHC, heptachlor, heptachlor epoxide, hexachlorobenzene, pentachlorophenol, TBT, and vanadium. Dioxins and furans were included as a COC for seafood consumption, even though no quantitative risk estimates were made. It was assumed that dioxin and furan excess cancer risk estimates for RME seafood consumption scenarios would have been greater than 1×10^{-6}.

[7] The COCs based on direct sediment exposure were PCBs, arsenic, cPAHs, dioxins and furans, and toxaphene.

drivers based on the magnitude of their risk estimates and the relative percentage of their contributions to total human health risk. PCBs, arsenic, cPAHs, and dioxins and furans were identified as risk drivers for both seafood consumption and direct sediment exposure scenarios. COCs not selected as risk drivers in the baseline HHRA will be addressed through focused evaluation in the FS; these chemicals may also be considered in remedial design for specific areas in or near the LDW, and included in the post-remedial monitoring program that is part of the 5-year review that EPA conducts at all Superfund sites where hazardous substances, pollutants, or contaminants remain in place after cleanup is completed.

These findings do not constitute a definitive characterization of human health risks. There are many uncertainties associated with the site-specific risk estimates for each exposure scenario. In spite of these uncertainties, the baseline risk characterization for the LDW site is considered to be health-protective and sufficient to support risk management decisions.

ES.6 PRELIMINARY BACKGROUND CONCENTRATIONS

Both EPA and Ecology recognize two types of background, natural and anthropogenic (also referred to as area), although their definitions and uses differ. The most important difference is in the potential application of anthropogenic background concentrations in making risk management decisions. EPA generally does not require cleanup to concentrations below anthropogenic background concentrations because of the potential for recontamination from sources unrelated to the site, cost effectiveness, and technical practicability (EPA 2002d). Under MTCA, natural background concentrations[8] are used in the evaluation of sediment cleanup levels as a lower limit below which cleanup levels cannot be achieved (WAC 173-340-705(6)).

Background information presented in the RI is intended to provide initial information and a preliminary context for the site characterization data and risk-based threshold concentrations (RBTCs) presented herein. This compilation is not exhaustive. Additional studies are underway and will provide new datasets relevant to LDW background determinations. Additional evaluation of background concentrations will be documented in the FS or in a separate technical memorandum before a ROD is completed.

Concentrations in surface sediment for four risk driver chemicals (total PCBs, arsenic, cPAHs, and dioxins and furans) were calculated using from data from Puget Sound reference areas (Table ES-7). These areas were designated as Puget Sound reference areas in various studies sponsored by federal and state agencies because they are in relatively undeveloped areas and represent areas outside the influence of urban bays

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[8] According to MTCA (WAC 173-340-200), natural background concentrations are concentrations of hazardous substances that are consistently present in an environment that has not been influenced by localized human activities.

and known point sources. The application of these Puget Sound reference area data to remedial decisions will be determined by EPA and Ecology.

Table ES-7. Concentrations of risk driver chemicals in samples collected from Puget Sound reference areas, upstream of the LDW, urban bays and lakes, and the Upper Turning Basin

		Surface Sediment Data			Subsurface Sediment Data	Upstream Water Data
Risk Driver Chemical	**Unit**	**Puget Sound Reference Areas[a]**	**Upstream[b]**	**Urban Bays and Lakes[c]**	**Upper Turning Basin[d]**	**Suspended Solids[e]**
Total PCBs	µg/kg dw	0.2 – 19.9	40	64 – 217	86	107
Arsenic	mg/kg dw	2.3 – 15.9	11	9.8 – 38	13	73
cPAHs[f]	µg/kg dw	14.7 – 244.7	135	185 – 904	201	354
Dioxin and furan TEQ	ng/kg dw	0.141 – 2.3	2.0[g]	16.3[h]	no data	no data

[a] Information relevant to preliminary natural background concentrations in sediment is presented in Section 7.1. The concentration ranges presented in this table (for informational purposes only) represent the ranges of 90th percentile concentrations in Puget Sound reference areas.

[b] Information relevant to upstream concentrations in sediment is presented in Section 7.2.2. The concentrations presented in this table (for informational purposes only) represent the 90th percentile concentrations in the upstream dataset.

[c] Information on concentrations from urban bays and lakes (excluding cleanup and disposal sites) is presented in Section 7.3 to provide additional context to the concentrations summarized in Sections 7.1 and 7.2. The concentration ranges presented in this table (for informational purposes only) represent the ranges of 90th percentile concentrations from the various datasets considered. Inner Elliott Bay data have been excluded from this summary because the bay receives discharge from the LDW and may be influenced by other known point sources of contamination.

[d] Information relevant to concentrations in subsurface sediment from the Upper Turning Basin is presented in Section 7.2.3. The concentrations presented in this table (for informational purposes only) represent the 90th percentile concentrations in sediment cores collected between RM 4.0 and RM 4.75.

[e] Information relevant to concentrations in suspended solids is presented in Section 7.2.4. For PCBs and cPAHs, suspended solids concentrations were estimated by normalizing whole-water concentrations in upstream samples to the sample-specific TSS concentrations. For arsenic, sample-specific total and dissolved concentrations and TSS in upstream water samples were used to estimate suspended solids concentrations. The concentrations presented in this table (for informational purposes only) represent the 90th percentile concentrations.

[f] cPAH concentrations are expressed in terms of benzo(a)pyrene equivalents.

[g] The TEQ presented for dioxins and furans represents the mean concentration of four upstream surface sediment samples.

[h] The TEQ presented for dioxins and furans represents the 90th percentile of the dataset collected near storm drains and other areas receiving surface runoff within the greater Seattle metropolitan area (Section 7.3).

cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw – dry weight
LDW – Lower Duwamish Waterway
MTCA – Model Toxics Control Act
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent
TSS – total suspended solids
Washington Administrative Code

The RI also presents surface sediment data from other areas outside of the LDW (e.g., Green River data upstream of the LDW and sediment data from urban bays and lakes), subsurface sediment data from the Upper Turning Basin, and suspended solids data from upstream of the LDW for the risk driver chemicals (Table ES-7). The RI does not recommend natural or anthropogenic background concentrations, which will be developed as part of the FS in consultation with EPA and Ecology.

ES.7 RISK-BASED THRESHOLD CONCENTRATIONS

Sediment RBTCs, which are defined as concentrations of risk driver chemicals in sediment that are associated with specific risks, are helpful in risk management decisions. If sediment were remediated to meet a sediment RBTC, for example, then the exposure conditions following that cleanup action would be associated with a reduction of risk down to the target risk level for that RBTC. RBTCs were estimated for each of the risk driver chemicals identified in the risk assessments. Sediment RBTCs were calculated for risks associated with direct sediment contact by people, seafood consumption, and protection of the benthic invertebrate community.

For each of the direct sediment contact RME scenarios (beach play, tribal clamming, and tribal netfishing), sediment RBTCs were calculated using the risk equation for three different excess cancer risk levels (Table ES-8). Other RBTCs were also calculated for informational purposes. For example, RBTCs associated with the central tendency scenarios were also calculated. Sediment RBTCs derived for human health direct-contact scenarios should be applied as SWACs over the exposure areas identified in the HHRA. For the netfishing scenario, the exposure area is the entire LDW. For the clamming and beach play scenarios, the exposure areas include intertidal areas where such activities may occur.

Table ES-8. Summary of sediment RBTCs for human health risk drivers for RME direct sediment exposure scenarios

RISK DRIVER CHEMICAL	UNIT	EXPOSURE SCENARIO	SEDIMENT RBTCS		
			1 IN 1,000,000 RISK LEVEL[a]	1 IN 100,000 RISK LEVEL[a]	1 IN 10,000 RISK LEVEL[a]
Arsenic	mg/kg dw	tribal clamming	1.3	13	130
		beach play	2.8	28	280
		netfishing	3.7	37	370
cPAHs[b]	µg/kg dw	tribal clamming	150	1,500	15,000
		beach play	90	900	9,000
		netfishing	380	3,800	38,000
Dioxin and furan TEQ	ng/kg dw	tribal clamming	13	130	1,300
		beach play	28	280	2,800
		netfishing	37	370	3,700
Total PCBs	µg/kg dw	tribal clamming	500	5,000	50,000
		beach play	1,700	17,000	170,000
		netfishing	1,300	13,000	130,000

Note: RBTCs were not calculated for non-cancer endpoints because estimated HQs were all < 1.

[a] Tribal clamming (120 days/yr).

[b] cPAH concentrations are expressed in terms of benzo(a)pyrene equivalents.

cPAH – carcinogenic polycyclic aromatic hydrocarbon RBTC – risk-based threshold concentration

dw – dry weight	RME – reasonable maximum exposure
HQ – hazard quotient	TEQ – toxic equivalent
PCB – polychlorinated biphenyl	

For seafood consumption, sediment RBTCs for total PCBs were estimated using a food web model. A range of RBTCs was calculated. Sediment RBTCs at the 1 in a million (1×10^{-6}) and 1 in 100,000 (1×10^{-5}) risk levels for the tribal RME (adult and child) scenario could not be calculated; the contribution of total PCBs from water alone was high enough to result excess cancer risks above those risk levels even in the absence of any contribution from sediment (the sediment RBTCs for these scenarios are expressed as < 1 µg/kg dw in Table ES-9). Sediment RBTCs for total PCBs for the non-RME scenarios ranged from < 1 µg/kg dw for the adult tribal scenario based on Suquamish data to > 380 µg/kg dw for a number of scenarios at the 1×10^{-4} target risk level (Table ES-10). A sediment RBTC of > 380 µg/kg dw indicates that even under current conditions in the LDW (i.e., the current LDW-wide SWAC is 380 µg/kg dw), excess cancer risks are estimated to be less than the target risk level.

Table ES-9. Sediment RBTCs for total PCBs based on excess cancer risks in HHRA RME seafood consumption scenarios

Scenario	Risk Level	Sediment RBTC (µg/kg dw)[a]		
		Lower Bound	Best Fit	Upper Bound
Adult tribal RME (Tulalip data)	1×10^{-4}	< 1[b]	7.3	25
Child tribal RME (Tulalip data)		109	185	301
API RME		67	100	167
Adult tribal RME (Tulalip data)	1×10^{-5}	< 1	< 1	< 1
Child tribal RME (Tulalip data)		< 1	< 1	< 1
API RME		< 1	< 1	< 1
Adult tribal RME (Tulalip data)	1×10^{-6}	< 1	< 1	< 1
Child tribal RME (Tulalip data)		< 1	< 1	< 1
API RME		< 1	< 1	< 1

[a] The best-fit RBTC was derived using the food web model parameter set that resulted in the closest match between empirical data and model estimates for all species. The lower and upper bounds represent RBTCs derived from the model based on the range of estimates from other parameter sets that passed the performance criterion (i.e., all model estimates within a factor of 2 of empirical data).

[b] RBTC could not be calculated because contribution from water alone resulted in estimated tissue concentrations greater than the applicable risk level, even in the absence of any contribution from sediment.

API – Asian and Pacific Islander	PCB – polychlorinated biphenyl
dw – dry weight	RBTC – risk-based threshold concentration
HHRA – human health risk assessment	RME – reasonable maximum exposure

Table ES-10. Sediment RBTCs for total PCBs based on excess cancer risks in the non-RME HHRA seafood consumption scenarios

SCENARIO	RISK LEVEL	SEDIMENT RBTC (µg/kg dw)[a]		
		LOWER BOUND	BEST FIT	UPPER BOUND
Adult tribal CT (Tulalip data)	1×10^{-4}	370	> 380	> 380
Child tribal CT (Tulalip data)		> 380	> 380	> 380
API CT		> 380	> 380	> 380
Adult tribal (Suquamish data)		< 1[b]	< 1	< 1
Adult one meal per month of benthic fish		220	320	> 380
Adult one meal per month of pelagic fish		170	250	> 380
Adult one meal per month of crabs		> 380	> 380	> 380
Adult one meal per month of clams		> 380	> 380	> 380
Adult tribal CT (Tulalip data)	1×10^{-5}	13	29	62
Child tribal CT (Tulalip data)		68	100	181
API CT		260	> 380	> 380
Adult tribal (Suquamish data)		< 1	< 1	< 1
Adult one meal per month of benthic fish		9	13	23
Adult one meal per month of pelagic fish		< 1	< 1	< 1
Adult one meal per month of crabs		100	220	> 380
Adult one meal per month of clams		270	350	> 380
Adult tribal CT (Tulalip data)	1×10^{-6}	< 1	< 1	< 1
Child tribal CT (Tulalip data)		< 1	< 1	< 1
API CT		0.2	10	28
Adult tribal (Suquamish data)		< 1	< 1	< 1
Adult one meal per month of benthic fish		< 1	< 1	< 1
Adult one meal per month of pelagic fish		< 1	< 1	< 1
Adult one meal per month of crabs		< 1	< 1	< 1
Adult one meal per month of clams		< 1	2	8

[a] The best-fit RBTC was derived using the food web model parameter set that resulted in the closest match between empirical data and model estimates for all species. The lower and upper bounds represent RBTCs derived from the model based on the range of estimates from other parameter sets that passed the performance criterion (i.e., all model estimates within a factor of 2 of empirical data).

[b] RBTC could not be calculated because contribution from water alone resulted in estimated tissue concentrations greater than the applicable risk level, even in the absence of any contribution from sediment.

API – Asian and Pacific Islander
CT – central tendency
dw – dry weight
HHRA – human health risk assessment
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure

At the 1 in 10,000 (1×10^{-4}) risk level, sediment RBTCs for total PCBs ranged from 7.3 to 185 µg/kg dw for the three RME scenarios (Table ES-9), which are lower than the sediment RBTCs at the 1 in a million (1×10^{-6}) risk level for the direct sediment exposure scenarios (500 to 1,700 µg/kg dw) (Table ES-8). The sediment RBTCs for total

PCBs calculated for river otter ranged from 128 to 159 µg/kg dw. These sediment RBTCs for total PCBs are lower than the current SWAC of total PCBs in the LDW (approximately 380 µg/kg dw).[9]

Sediment RBTCs were not calculated for arsenic, cPAHs, or dioxins and furans for the human seafood consumption pathway because either data were not available or the relationship between sediment and tissue for these chemicals is not sufficiently understood (Section 8.3). Sediment RBTCs for the protection of benthic invertebrates were the SQS and CSL from the SMS.

A comparison of sediment RBTCs to background concentrations will be important in risk management decisions made by EPA and Ecology because cleanup to concentrations below background concentrations is not practicable. The comparison of RBTCs with preliminary background datasets presented here uses the 90th percentile of each of the background datasets. Sediment RBTCs at the 1×10^{-6} risk level for direct sediment contact are higher than the range of Puget Sound reference area concentrations for total PCBs and dioxins and furans (Table ES-11). The cPAH sediment RBTC for netfishing at the 1×10^{-6} risk level is also greater than the range of Puget Sound reference area concentrations. The cPAH direct-contact sediment RBTCs at the 1×10^{-6} risk level for beach play and clamming are within the range of Puget Sound reference area concentrations. For arsenic, the direct sediment contact RBTCs at the 1×10^{-6} risk level are toward the lower end or below the range of the Puget Sound reference area concentrations.[10] At higher risk levels, direct-contact sediment RBTCs are above the concentration ranges of samples collected from Puget Sound reference areas, upstream, and the Upper Turning Basin, except for arsenic. The direct-contact sediment RBTCs for tribal clamming for arsenic at the 1×10^{-5} risk level is within the range of these concentrations.

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[9] Using the inverse distance weighting (IDW) interpolation method documented in ENSR (2008), the SWAC of total PCBs for the LDW from RM 0.0 to 5.25 was 350 µg/kg dw. Based on an earlier IDW interpolation used in the food web model (FWM), the SWAC was 380 µg/kg dw for the LDW from RM 0.0 to RM 5.25. The sensitivity of the FWM to this difference in SWAC was evaluated as part of the sensitivity and uncertainty analysis of the model (Appendix D), and was found to be low.

[10] Of the 10 Puget Sound reference areas, one area (Rich Passage) had a 90th percentile arsenic concentration of 2.3 mg/kg dw; the remaining nine areas had 90th percentile arsenic concentrations ranging from 6.3 to 15.9 mg/kg dw.

Table ES-11. Comparison of sediment RBTCs and concentrations for selected risk driver chemicals in samples collected from Puget Sound reference areas, upstream of the LDW, and the Upper Turning Basin

RISK DRIVER CHEMICAL	Unit	RISK LEVEL	SEDIMENT RBTCs FOR RME DIRECT-CONTACT SCENARIOS (TRIBAL CLAMMING, BEACH PLAY, NETFISHING)	SEDIMENT RBTCs FOR RME SEAFOOD CONSUMPTION SCENARIOS	SURFACE SEDIMENT DATA		SUBSURFACE SEDIMENT DATA	WATER DATA
					PUGET SOUND REFERENCE AREAS[a]	UPSTREAM[b]	UPPER TURNING BASIN[c]	UPSTREAM SUSPENDED SOLIDS[d]
Arsenic	mg/kg dw	1×10^{-6}	1.3, 2.8 ,3.7[e]	nc[f]	2.3 – 15.9	11	13	73
		1×10^{-5}	13 ,28, 37[e]	nc[f]				
		1×10^{-4}	130, 280, 370[e]	nc[f]				
		HQ = 1	na	na				
		SQS	57	na				
		CSL	93	na				
cPAHs[g]	µg/kg dw	1×10^{-6}	150, 90, 380[e]	nc[h]	14.7 – 244.7	135	201	354
		1×10^{-5}	1,500, 900, 3,800[e]	nc[h]				
		1×10^{-4}	15,000, 9,000, 38,000[e]	nc[h]				
		HQ = 1	na	na				
Dioxin and furan TEQ	ng/kg dw	1×10^{-6}	13, 28, 37[e]	nc[i]	0.141 – 2.3	2.0[j]	nd	nd
		1×10^{-5}	130, 280, 370[e]	nc[j]				
		1×10^{-4}	1,300, 2,800, 3,700[e]	nc[j]				
		HQ = 1	na	na				
Total PCBs	µg/kg dw	1×10^{-6}	500, 1,700, 1,300[e]	< 1[k]	0.2 – 19.9	40	86	107
		1×10^{-5}	5,000, 17,000, 13,000[e]	< 1[k]				
		1×10^{-4}	50,000, 170,000, 130,000[e]	7.3 – 185[l]				
		HQ = 1	nc[m]	128 – 159[n]				
		SQS	130[o]	na				
		CSL	1,000[p]	na				

[a] Information relevant to calculating preliminary natural background concentrations in sediment is presented in Section 7.1. The concentration ranges presented in this table (for informational purposes only) represent the ranges of 90th percentile concentrations calculated for multiple Puget Sound reference areas.

[b] Information relevant to upstream concentrations in sediment is presented in Section 7.2. The concentrations presented in this table (for informational purposes only) represent the 90th percentile concentrations in the upstream and Upper Turning Basin datasets. Other anthropogenic inputs from the surrounding urban area are likely due to atmospheric deposition and stormwater runoff but were not quantified as part of the RI studies. Concentrations in urban bays and lakes are discussed in Section 7.3.

[c] Information relevant to concentrations in subsurface sediment from the Upper Turning Basin is presented in Section 7.2.3. The concentrations presented in this table (for informational purposes only) represent the 90th percentile concentrations in sediment cores collected between RM 4.0 and RM 4.75.

[d] Information relevant to concentrations in suspended solids is presented in Section 7.2.4. For PCBs and cPAHs, suspended solids concentrations were estimated by normalizing whole-water concentrations in upstream samples to the sample-specific total TSS concentrations. For arsenic, sample-specific total and dissolved concentrations and TSS in upstream water samples were used to estimate suspended solids concentrations. The concentrations presented in this table (for informational purposes only) represent the 90th percentile concentrations.

[e] Sediment RBTCs developed from the human health direct-contact RME scenarios (i.e., tribal clamming/beach play/netfishing).

[f] Although identified as a COC for seafood consumption, sediment RBTCs were not calculated for arsenic for seafood consumption scenarios because a clear relationship has not been established between arsenic concentrations in surface sediment and clam tissues, but would be needed to develop an RBTC.

[g] cPAH concentrations are expressed in terms of benzo(a)pyrene equivalents.

[h] Although identified as a COC for seafood consumption, sediment RBTCs were not calculated for cPAHs for seafood consumption scenarios because a clear relationship has not been established between cPAH concentrations in surface sediment and clam tissues, but would be needed to develop an RBTC.

[i] Although identified as a COC for seafood consumption, sediment RBTCs were not calculated for dioxins and furans for seafood consumption scenarios because no tissue data were available from the LDW at the time of the risk assessments.

[j] Because of the small dataset, 90th percentiles were not calculated for dioxin and furan TEQs. The dioxin and furan TEQ represents the mean TEQ in upstream surface sediment samples.

[k] A sediment RBTC could not be calculated; even if the total PCB concentration in sediment was set equal to 0 µg/kg dw, FWM-estimated total PCB concentrations in tissue would be greater than the tissue RBTC for the applicable risk level because of the contribution of PCBs from water alone.

[l] Sediment RBTCs developed from the human health seafood consumption RME scenarios.

[m] Sediment RBTCs were not estimated for non-cancer hazards for direct-contact scenarios because none of the RME scenarios had HQs for an individual chemical greater than 1 or generated endpoint-specific HIs in excess of 1 (Appendix B, Section B.5.6).

[n] RBTCs developed from the otter prey ingestion scenario, estimated through the FWM.

[o] Reported value is lowest AET, which is functionally equivalent to the SQS, but in dry weight units.

[p] Reported value is second lowest AET, which is functionally equivalent to the CSL but in dry weight units.

AET – apparent effects threshold
COC – chemical of concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CSL – cleanup screening level
dw – dry weight
FWM – food web model
HI – non-cancer hazard index
HQ – non-cancer hazard quotient
LDW – Lower Duwamish Waterway
na – not available or applicable
nc – not calculated
nd – no data
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RI – remedial investigation
RM – river mile
RME – reasonable maximum exposure
SQS – sediment quality standard
TEQ – toxicity equivalent quotient
TSS – total suspended solids

Tissue RBTCs for total PCBs were also calculated for a given risk threshold and seafood consumption rate. For example, the total PCB concentration in seafood that would be necessary to achieve an excess cancer risk of 1 × 10^{-5} would be 4.2 µg/kg ww for the adult tribal RME scenario and 14 µg/kg ww for the API RME scenario. Just as it is important to put sediment RBTCs in the context of sediment background levels, it is important to put tissue RBTCs in the context of the PCB concentrations in other common dietary food items. As described in Section 7.4.1 and in WSDOH (2006), PCBs are frequently detected in many different seafood species throughout Puget Sound at concentrations similar to these tissue RBTCs (Table ES-12). Mean concentrations in Puget Sound seafood ranged from 4.5 to 32.3 µg/kg ww in rock fish, 2.8 to 11.6 µg/kg ww in English sole fillet, 0.62 to 25 µg/kg ww in crab edible meat and hepatopancreas, and 0.12 to 2.8 µg/kg ww in bivalves (WSDOH 2006). In addition, these tissue RBTCs are lower than the PCB concentrations that have been detected in some common food items from a typical diet (WSDOH 2006). Consumption of many of these common food items at the rates assumed under these RME scenarios would subject the consumer to unacceptable risks.

Table ES-12. Total PCB concentrations in LDW and greater Puget Sound tissue compared with tissue RBTCs

EPC OR RBTC	TOTAL PCB CONCENTRATION (µg/kg ww)		
	ENGLISH SOLE	CLAM	DUNGENESS AND SLENDER CRAB
LDW-wide 95% UCL	1,200	600	200
LDW-wide mean	700	140	170
Greater Puget Sound (range of mean values)[a]	2.8 – 11.6[b]	0.12 – 2.8[c]	0.62 – 8.44[d]
1 × 10^{-4} RBTC (Tulalip RME)	42	42	42
1 × 10^{-5} RBTC (Tulalip RME)	4.2	4.2	4.2
1 × 10^{-6} RBTC (Tulalip RME)	0.42	0.42	0.42

[a] Range of mean total PCB concentrations from various studies and non-urban sampling areas in the greater Puget Sound area, as presented in Table 7-19.

[b] Total PCB concentrations in English sole fillet tissue.

[c] Total PCB concentrations in clam soft tissue.

[d] Total PCB concentrations in crab edible meat or calculated whole-body tissue.

EPC – exposure point concentration
LDW – Lower Duwamish Waterway
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
RBTC – risk-based threshold concentration
UCL – upper confidence limit on the mean
ww – wet weight

ES.8 POTENTIAL CHEMICAL SOURCES AND ONGOING SOURCE CONTROL EFFORTS

Much of the sediment contamination in the LDW is believed to be the result of historical sources. Many of these historical sources, including direct discharges of municipal and industrial wastewater, have largely been controlled through compliance with modern regulatory requirements, improved business practices, elimination of product use, or cleanups of industrial property. Many of the historical direct discharges have been diverted to the sanitary sewer system and are now treated at the West Point wastewater treatment plant. Inspection and compliance programs are in place to monitor and control the handling and disposal of manufacturing wastes produced by industrial operations along the LDW. In addition, the chemical loading from stormwater and combined sewer overflows (CSOs) has been greatly reduced by permit programs that require the monitoring and control of discharges that could otherwise enter the LDW.

Although sources of chemicals to the LDW are much reduced in number and magnitude compared with the past, as chemical profiles in sediment cores demonstrate, sources that could lead to sediment contamination, or recontamination of sediment following cleanup, still exist in some areas of the LDW. Ecology, in its lead role in the Source Control Work Group (SCWG), is responsible for investigating these potential ongoing sources and evaluating whether they have been controlled sufficiently for sediment cleanup to begin.

Potential ongoing sources include stormwater runoff, CSOs, industrial wastewater discharges, deposition from uncontrolled or partially controlled air emissions (from motor vehicles and the burning of wood and fossil fuels), illicit discharges and spills, erosion of contaminated bank material, groundwater, and upstream contributions from the Green River. Ongoing investigations are being conducted at several facilities near the LDW.

The characterization of potential chemical sources to the LDW began in the 1980s. In recent years, considerable effort has been expended in sampling potential upland sources. The chemicals most often detected in these samples are PCBs, TPH, phthalates, and metals (i.e., copper, mercury, and zinc).

Source control efforts will continue until all areas with significant potential to contaminate the LDW have been addressed. Ecology and the SCWG have identified 23 nearly contiguous areas within the LDW as an organizational tool to prioritize source investigations. Seven of these areas were previously identified as candidate EAAs based on sediment contamination (Windward 2003e). Removal actions have been implemented at two of these seven locations (i.e., Duwamish/Diagonal and Norfolk) and are currently in the planning stages for three more (Slip 4, Boeing Plant 2/Jorgensen Forge, and Terminal 117).

Initial source control efforts have been focused on source identification at the five EAAs where removal actions have occurred or are being planned. Current and future source control efforts include compiling summaries of existing information, identifying data gaps, and preparing source control action plans for the identified source control areas. These efforts help direct future investigation and remediation efforts while assessing potential sources of contamination to LDW sediments.

ES.9 KEY OBSERVATIONS AND FINDINGS

Key observations and findings for the RI are summarized below.

- Over the past 100 years, the LDW has been highly modified from its natural configuration to support urban and industrial development. Changes have included reductions and control of water flow, significant shoreline modifications, loss of intertidal habitat, and installation of riprap, pier aprons and sheet pile walls. Some limited areas of natural shoreline still exist within the LDW.
- Industrial and commercial facilities occupy most of the shoreline; one residential community is also located along the shoreline with another community nearby.
- The LDW is currently used as an industrial navigational corridor. It also supports recreational uses such as boating, kayaking, fishing, and beach play. The LDW is also one of the locations of the Muckleshoot Tribe's commercial, ceremonial, and subsistence fishery for salmon, and the Suquamish Tribe actively manages aquatic resources north of the Spokane Street Bridge, located just north of the LDW. The Duwamish Tribe uses Herring's House Park and other parks along the Duwamish for cultural gatherings.
- Despite significant alterations in habitat and areas with elevated chemical concentrations, the LDW contains a diverse assemblage of aquatic and wildlife species and a robust food web that includes top predators.
- The majority of the high arsenic and total PCB concentrations in surface sediment were located within fairly well-defined areas. The locations of the highest arsenic and total PCB concentrations were generally not in the same areas, indicating that sources likely differ for these two chemicals. Areas with the highest cPAH concentrations were located in many of the same areas identified for arsenic and total PCBs, but were also more dispersed. There are several areas with high dioxin and furan TEQs in surface sediments.
- Most of the human health risk is from PCBs, arsenic, cPAHs, and dioxins and furans.
- The highest risks to people are associated with consumption of fish, crabs, and clams, with lower risks associated with activities that involve direct contact with sediment, such as clamming, beach play, and netfishing.
- Ecological risks to fish and wildlife were relatively low, with the exception of risks to river otter from PCBs.

- Based on sediment chemistry and toxicity test results, sediment contamination in approximately 75% of the LDW (345 ac) is estimated to have no effect on the benthic invertebrate community; approximately 7% (34 ac) of the surface sediment has chemical concentrations exceeding the higher of the two state standards, and is therefore expected to have adverse effects on the benthic invertebrate community (see WAC 173-204-310[b]). The remaining 18% of the LDW (82 ac) has chemical concentrations exceeding the lower of the two state standards and these areas are considered to have minor adverse effects on the benthic invertebrate community. Most of the exceedances of SMS criteria were for PCBs and phthalates, although 41 different chemicals had at least one exceedance.
- Sediment is continually depositing within the LDW, with almost all new sediment (99%) originating from the Green River. The STM estimates that over 200,000 metric tons of sediment per year enter the LDW. Approximately 50% of this load deposits in the LDW. STM modeling runs indicate that approximately 90% of the total bed area in the LDW receives 10 cm of new sediment (from the combined Green River and lateral sources) within 10 years or less. This sediment is mixed with the existing bedded sediment through various processes, including bioturbation and propeller wash.
- A few areas in the LDW will be scoured during high-flow events. Based on the STM, the maximum scour depth is relatively shallow and is generally limited to sediment in the top 20 cm; thus, deeper sediments would not be exposed as a result of high-flow events. Scour to these relatively shallow depths is expected to occur in relatively small areas of the LDW. The STM did not account for scour from localized activities, such as discharges from outfalls, tugboat maneuvering, or anchor dragging, that could have caused localized erosional environments. Routine boat traffic is expected to mix the top few cm of sediment, which is part of the biologically active zone mixed by benthic invertebrates, whereas tugboat maneuvering is a potential source of localized erosion that could disturb sediment at greater depths in small areas. In addition, in some areas, ships may have caused localized erosion from physical forces (i.e., anchor dragging) unrelated to propeller-driven scour. Site-specific information, in addition to the results of the STM, will be evaluated in any future remedial designs.
- The physical conceptual site model of a net depositional environment is supported by both physical and chemical lines of evidence, including lithology and chemistry profiles in sediment, where these data were collected. Empirically derived net sedimentation rates from chemical and physical markers were within 1 cm/yr of the modeled sedimentation rates for 40 of

63 core locations. Most (70%)[11] of the 157 core locations evaluated had depths of peak PCB concentrations that were consistent with those predicted by the STM, with peak PCB concentrations as deep or deeper than those estimated by the STM. The remaining 30% of the cores had peak concentrations that were shallower than expected, assuming net sedimentation rates from the STM, peak PCB usage and release in the 1960s and 1970s, and minimal localized disturbances.

- Based on the STM, LDW surface sediment is generally expected to become more similar in character over time to the sediment being transported by the Green River on an LDW-wide scale; localized areas may continue to be influenced by inputs from sources in those areas.
- A comparison of background concentrations with risk-based goals in sediment (represented by sediment RBTCs) will be important in risk management decisions by EPA and Ecology. Puget Sound sediment data will be considered in the derivation of natural background concentrations, and data from other lines of evidence will be considered in the derivation of anthropogenic background concentrations.
- For total PCBs, all sediment RBTCs for direct-contact scenarios are above the concentration range of samples collected from Puget Sound reference areas, upstream of the LDW, and from the Upper Turning Basin (subsurface sediment). All sediment RBTCs for RME seafood consumption scenarios are below the range of Puget Sound reference area concentrations across the risk range of 1×10^{-6} to 1×10^{-4}, except for three sediment RBTCs. Sediment RBTCs for the child Tribal RME and the API RME scenarios (at 1×10^{-4} risk level) and for the protection of river otters are within the range or higher than the concentration range of samples collected from upstream of the LDW and from the Upper Turning Basin (subsurface sediment).
- No sediment RBTCs were derived for seafood consumption scenarios for arsenic, cPAHs, and dioxins and furans. Sediment RBTCs for arsenic and cPAHs for seafood consumption could not be estimated because the relationships between concentrations in clam tissue and sediment are not well understood and are highly uncertain. Sediment RBTCs for dioxins and furans for seafood consumption could not be estimated because there were no tissue data from the LDW when the risk assessments were conducted.
- For arsenic, sediment RBTCs for direct-contact scenarios at the 1×10^{-6} risk level are below the concentration range of samples collected from upstream of the

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[11] The percent of cores with peak concentrations as deep or deeper than expected is 65% when 24 cores were excluded from the analysis because they had samples composited over intervals ≥ 3 ft in the portion of the core where the peak PCB concentrations were estimated or detected.

LDW and from the Upper Turning Basin (subsurface sediment) and are just below or within the wider range of concentrations in Puget Sound reference area samples. The tribal clamming direct-contact sediment RBTC at the 1 × 10^{-5} risk level is above the upstream concentration and the same as from the Upper Turning Basin (subsurface sediment) concentration but within the range of concentrations in Puget Sound reference area samples. The beach play and netfishing direct-contact sediment RBTCs at the 1 × 10^{-5} risk level are above concentrations found in Puget Sound reference areas, upstream of the LDW, and the Upper Turning Basin. Direct-contact sediment RBTCs at the 1 × 10^{-4} risk level are above all of these concentrations.

- Sediment RBTCs for cPAHs for the tribal clamming and beach play direct-contact scenarios at the 1 × 10^{-6} risk level are within the concentration ranges of samples collected from Puget Sound reference areas, upstream of the LDW, and from the Upper Turning Basin (subsurface sediment). The netfishing sediment RBTC for cPAHs at the 1 × 10^{-6} risk level is above all of the aforementioned concentration ranges as are all direct-contact sediment RBTCs at the 1 × 10^{-5} and 1 × 10^{-4} risk levels.
- Sediment RBTCs for dioxins and furans for direct-contact scenarios across the excess cancer risk range of 1 × 10^{-6} to 1 × 10^{-4} are above the concentration ranges of samples collected from Puget Sound reference areas and upstream of the LDW.

1 Introduction

This document presents the results of the remedial investigation (RI) for the Lower Duwamish Waterway (LDW). Located in Seattle, Washington, the LDW consists of the downstream portion of the Duwamish River, excluding the East and West Waterways around Harbor Island, and has served as the City of Seattle's major industrial corridor since about 1911. The area sampled as part of the RI extends from River Mile (RM) 0.0, near the southern tip of Harbor Island, to upstream of the Upper Turning Basin to approximately RM 6.0 (Map 1-1).

The LDW has been the subject of numerous environmental investigations and characterization studies completed under the auspices of various governmental and private entities, including the US Environmental Protection Agency (EPA), the National Oceanic and Atmospheric Administration (NOAA), The Boeing Company (Boeing), King County, the Port of Seattle, the Elliott Bay/Duwamish River Restoration Program, and the US Army Corps of Engineers (USACE). The LDW was evaluated and proposed for inclusion on the National Priorities List (NPL) pursuant to Section 105 of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, or Superfund), 42 US Code 9605, by EPA; and on September 13, 2001, the LDW was formally listed as a Superfund site. In February 2002, the Washington State Department of Ecology (Ecology) listed the LDW under the authority of the Washington State Model Toxics Control Act (MTCA).

Superfund and MTCA regulations require that an RI and a feasibility study (FS) be conducted for all listed sites. In 2000, the City of Seattle, King County, the Port of Seattle, and Boeing, working collectively as the Lower Duwamish Waterway Group (LDWG), agreed in an Administrative Order on Consent (AOC) to conduct an RI/FS for the LDW. Oversight of the LDW RI/FS is being provided by EPA and Ecology. As part of the RI, existing data were synthesized and new data were analyzed to determine the nature and extent of contamination, assess sources of contamination to the LDW, and conduct baseline risk assessments. Information collected during the RI will be used to help identify areas requiring cleanup. The purpose of the FS is to analyze and compare alternatives or approaches to remediate those areas that need cleanup to reduce risks.

1.1 LDW RI PROGRAM OVERVIEW

To maximize the effective use of existing data, the LDW RI was completed in two phases (Windward 2000). For the Phase 1 RI (Windward 2003a), LDWG compiled and analyzed existing data to support an understanding of the nature and extent of contamination in the sediments of the LDW and presented preliminary risk estimates based on those chemical distributions. Information evaluated during the Phase 1 RI was used to support the identification of candidate early remedial action sites also known as early action areas (EAAs). EAAs were selected using a risk-based approach to identify high-priority areas for which remedial actions could be initiated prior to completion of the RI/FS process, as described in a technical memorandum submitted to EPA and Ecology in July 2003 (Windward 2003e). The Phase 1 RI also served as the basis for identifying additional data needed to complete the RI and the baseline ecological risk assessment (ERA) and human health risk assessment (HHRA) (Windward 2003f).

The principal objective of Phase 2 of the RI process was to collect and evaluate data necessary to support the RI/FS and future risk management decision making for the site. Phase 2 tasks included:

- Further characterize the physical and biological nature of the LDW area, including human use
- Further characterize the nature and extent of contamination within the LDW area
- Summarize available source information and the status of source control activities
- Estimate risks to both human health and the environment under baseline conditions (prior to early action) and identify risk driver chemicals
- Determine risk-based threshold chemical concentrations in sediment for these risk drivers

To complete these tasks, additional studies and investigations were conducted as follows:

- Tissue studies
    - Juvenile chinook salmon tissue sampling and chemical analyses
    - Fish, clam, and crab tissue sampling and chemical analyses
    - Benthic invertebrate community characterization, including co-located tissue and sediment sampling and chemical analyses
- Clam, crab, and shrimp surveys
- Groundwater/porewater studies
    - Groundwater seep survey and chemical analyses
    - Sediment porewater sampling and chemical analyses
- Sediment studies
    - Sediment transport studies
    - Surface sediment sampling, chemical analyses, and toxicity testing
    - Subsurface sediment sampling and chemical analyses

- Other supporting studies
    - Reconnaissance studies to assess site use by sandpiper
    - Reconnaissance studies to assess potential human use of the LDW shoreline by means of various public access points
    - Gastropod study to assess imposex
    - Bathymetry study

The protocols and study design of these investigations were developed in consultation with EPA and Ecology. Results have been discussed in various documents and technical memoranda and reviewed by various public stakeholders; all results are available at www.ldwg.org/rifs_docs.htm.The results of these investigations, along with acceptable Phase 1 RI data and other new data of acceptable quality, have been used in the development of the baseline ERA and HHRA and are incorporated in this RI report.

1.2 DOCUMENT ORGANIZATION

This document is organized as follows:

- Section 1 – Introduction
- Section 2 – Environmental Setting
- Section 3 – Sediment Dynamics
- Section 4 – Nature and Extent of Contamination
- Section 5 – Summary of the Baseline Ecological Risk Assessment
- Section 6 – Summary of the Baseline Human Health Risk Assessment
- Section 7 – Preliminary Background Concentrations of Risk Driver Chemicals
- Section 8 – Risk-Based Threshold Concentrations
- Section 9 – Potential Pathways, Source Identification, and Source Control Efforts
- Section 10 – Key Findings
- Section 11 – References

These sections are supported by the following appendices:

- Appendix A – Baseline Ecological Risk Assessment
- Appendix B – Baseline Human Health Risk Assessment
- Appendix C – Implications of Incorporating Round 3 Surface Sediment Data in Risk Assessments
- Appendix D – Food Web Model

- Appendix E – Data Selection for the RI Baseline Surface Sediment Dataset, Data Quality Review Summaries, Data Management Rules, Additional Statistical Information, Summary Data Tables, and Arsenic and cPAH Concentrations in Sediment Associated with Tissue RBTCs
- Appendix F – Subsurface Core Analysis
- Appendix G – Listed Properties and Source Control Documentation
- Appendix H – LDW Outfall Locations
- Appendix I – Source Control Area-Related Facility Information
- Appendix J –Boring Logs Used for the LDW Hydrostratigraphy Evaluation

In addition, oversized maps are bound in a separate map folio. A CD containing the complete dataset used in this RI is attached to the back cover. In addition, a separate dataset is included on the CD that contains all of the trumped surface sediment data.

2 Environmental Setting

This section describes the LDW, its history, and its environmental setting, including physiography, physical characteristics, hydrogeology, ecological habitats, biological communities, and land use.

2.1 SITE DESCRIPTION

The Duwamish River originates at the confluence of the Green and Black Rivers (Map 2-1), and flows northwest for approximately 19 km (12 mi), splitting at the southern end of Harbor Island to form the East and West Waterways, prior to discharging into Elliott Bay. The LDW study area consists of the downstream portion of the Duwamish River, excluding the East and West Waterways (Map 2-1).[12]

The LDW serves as a major shipping route for bulk and containerized cargo, and the shoreline along the majority of the LDW has been developed for industrial and commercial operations. Common shoreline features include constructed bulkheads, piers, wharves, sheet piling walls, buildings that extend over the water, and steeply sloped banks armored with riprap or other fill material (Weston 1999a). A portion of the LDW (i.e., the reach downstream of the Upper Turning Basin) is maintained as a federal navigation channel by USACE (Weston 1999a). Authorized navigation channel depths are -30 ft mean lower low water (MLLW) from Harbor Island to the 1st Avenue S Bridge to -20 ft MLLW from the 1st Avenue S Bridge to Slip 4 and –15 ft MLLW from Slip 4 to the Upper Turning Basin (NOAA 2009b). Intermittent, shallow benches exist in the nearshore, intertidal, and shallow subtidal zones of the LDW, outside the navigation channel. These benches are of various dimensions and elevations, with minimum elevations of less than 3 ft MLLW (Windward and QEA 2008). A shallow sill is present at the southern end of the East Waterway, and a deeper sill is present at the southern end of the West Waterway, where the LDW splits around Harbor Island (Weston 1999b).

Intertidal habitats are dispersed in relatively small patches (i.e., generally less than 1 acre [ac] in size), with the exception of Kellogg Island, which represents the largest contiguous area of intertidal habitat remaining in the Duwamish River (Tanner 1991). Areas of low intertidal mudflats are present below upper bank riprap in the reaches upstream of the 1st Avenue S Bridge.

The LDW is used as an industrial and commercial corridor, consistent with the land use, zoning, and land ownership in the LDW. The LDW is also part of Tribal Usual and Accustomed (U&A) fishing areas. The Muckleshoot Indian Tribe currently conducts seasonal netfishing operations in the LDW for commercial, ceremonial, and

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[12] The East and West Waterways are part of the Harbor Island Superfund site.

subsistence purposes. The Suquamish Tribe actively manages resources north (downstream) of the Spokane Street Bridge, located just north of the LDW. The Duwamish Tribe uses Herring's House Park and other parks along the Duwamish for cultural gatherings.

Two City of Seattle neighborhoods, South Park and Georgetown, are located near the LDW. Both neighborhoods include a mixture of residential, recreational, commercial, and industrial land uses. The South Park neighborhood is located on the west side of the LDW, near the southern border of the City of Seattle's city limits. A portion of the South Park residential area abuts the LDW, and several houses are located along the shoreline. The Georgetown neighborhood is located east of the LDW and E Marginal Way S. The LDW also supports various recreational activities, including boating, kayaking, fishing, and beach play.

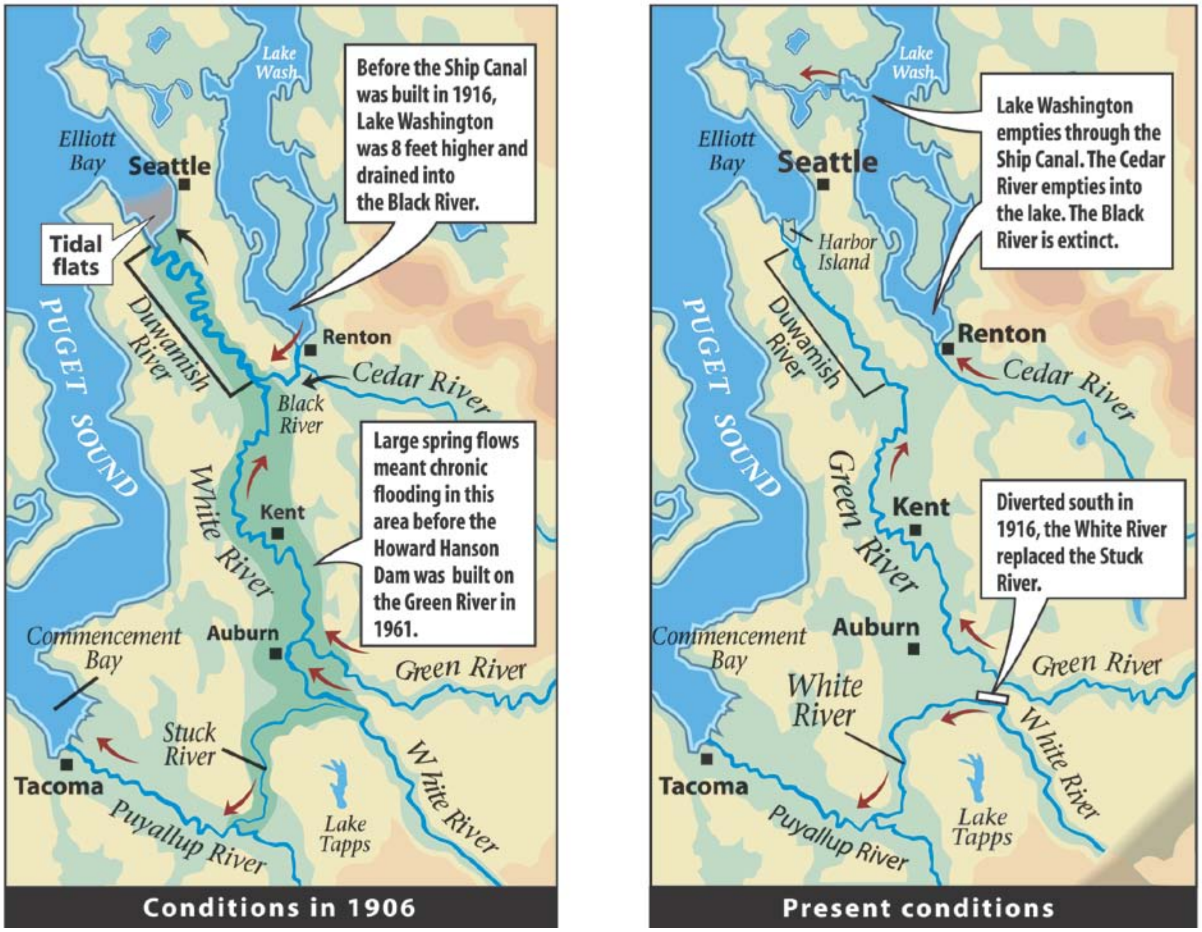
Interest in restoring and improving the intertidal habitat and other uses has led to several restoration projects, including the creation of public parks. Many restoration projects have been completed, and efforts are expected to continue over time (see Sections 2.8.1 and 2.9.3.4). Efforts to improve street ends and access points have also taken place (DRCC and ECOSS 2008; NOAA 2007b; Port of Seattle 1985, 1992). A more detailed description of site use is presented in Section 2.8.

2.2 SITE HISTORY

Prior to the 20th century, the Duwamish River was fed by the Green, Black, and White Rivers, with a combined drainage area of approximately 4,250 km^2 (1,640 mi^2) (Blomberg et al. 1988). The river meandered through a valley of floodplains, freshwater wetlands, and tidal marshes before emptying into Elliott Bay (Map 2-1). Flooding was a common occurrence in the river valley. Prior to the 1850s, the Duwamish River area was occupied by Native American tribal communities. The primary activities of these native peoples included fishing, hunting, gathering, and some limited farming, and some of the activities, notably fishing, continue to the present day. People of European origin began to settle in the area around the 1850s, clearing the Duwamish shorelines and draining wetlands for logging and agricultural purposes. In the early 1900s, continued issues with flooding in the area precipitated the installation of levees and dams, and the subsequent channelization of the river. As the quality of transportation and roads improved, commercial, industrial, and residential developments were constructed in the cleared lowland forest and farmland areas.

Throughout the 1900s, the watershed area and flow volumes to the Duwamish River were reduced by about 70% as a result of the diversion of the river's tributaries (Kerwin and Nelson 2000) (Figure 2-1). In 1906, the White River was diverted to the Puyallup River to help control flooding (Harper-Owes 1983). In 1916, the Black River, which was fed by the Cedar River and Lake Washington, was reduced to a minor stream when the level of Lake Washington was lowered through the construction of the Lake Washington Ship Canal, and the Cedar River was subsequently diverted to

Lake Washington (Harper-Owes 1983). Today, the Green River is the primary source of water for the Duwamish River.



Conditions in 1906

Before the Ship Canal was built in 1916, Lake Washington was 8 feet higher and drained into the Black River.

Large spring flows meant chronic flooding in this area before the Howard Hanson Dam was built on the Green River in 1961.

Present conditions

Lake Washington empties through the Ship Canal. The Cedar River empties into the lake. The Black River is extinct.

Diverted south in 1916, the White River replaced the Stuck River.

Source: Original artwork by Ben Garrison. Image provided courtesy of WRIA 9 Forum of Local Governments and King Conservation District.

Figure 2-1. Historical changes to the Duwamish River system

To facilitate navigation and economic development, the LDW was straightened and dredged in many areas by Commercial Waterway District No. 1 (Washington Statute Revised Code of Washington [RCW] 91.04). Dredging between 1903 and 1905 created the East and West Waterways, and dredged material from the river was used to create Harbor Island (Weston 1993a). The river has been dredged and channelized from just upstream of the Upper Turning Basin to the southern tip of Harbor Island since about 1916. Upstream of the Upper Turning Basin, the river is contained by dikes.

Most of the upland areas adjacent to the LDW have been heavily industrialized for many decades. Historical and current commercial and industrial operations include cargo handling and storage, marine construction, boat manufacturing, marina operations, concrete manufacturing, paper and metals fabrication, food processing, and airplane parts manufacturing. Two mixed commercial and residential communities, Georgetown and South Park, are also located near the waterway (Wilma 2001a).

The arrival of Seattle's first railroad in 1874 attracted numerous industrial activities. All major rail connections to Seattle from the south ran through Georgetown. Early tracks crossed the mudflats on pilings; and, in later years, railroad yards were built on fill dredged from the river. Industrial development increased as the mudflats were filled with soil from Seattle's former hills. In 1928, Seattle's first municipal airport, Boeing Field, was opened. Seven years later, Boeing opened its Plant 2 on the west side of Boeing Field (Wilma 2001a).

The neighborhood of South Park is located on the west bank of the LDW. Established as a farming community, South Park grew by more than 66 ac when the Duwamish was straightened, and the new land was claimed by farmers. In the 1920s, Boeing, located just north of South Park, built airplanes for the military. During World War II, South Park changed rapidly as industry from neighboring Georgetown attracted workers, creating a critical housing shortage in the farming community (Wilma 2001b).

2.3 PHYSIOGRAPHY

The LDW is located at the downstream end of the 1,466-km^2 (566-mi^2) Green/Duwamish River watershed (referred to as Water Resource Inventory Area [WRIA] 9), which includes portions of the Cities of Seattle, Tukwila, SeaTac, Renton, Kent, Federal Way, Auburn, Black Diamond, and Enumclaw, plus forested areas in unincorporated south-central King County (Map 2-2).

From the south end of Harbor Island to just south of the Upper Turning Basin, the LDW is approximately 8.8 km (5.5 mi) in length. The depth of the LDW varies from approximately -56 ft MLLW near the mouth to -10 ft MLLW near the head of the navigation channel (RETEC 2006). The average width of the LDW is 134 m (440 ft), widening downstream of the 1st Avenue S Bridge. The width of the navigation channel is 45.7 m (150 ft) (RETEC 2006).

2.3.1 LDW bathymetry

Numerous bathymetric surveys have been conducted in the LDW, but the most comprehensive survey was conducted in 2003 as part of the LDW RI (Windward and DEA 2004). The objective of the 2003 high-resolution, multibeam, bathymetric survey was to produce a bank-to-bank (where possible) bathymetric dataset for the LDW study area. The multibeam data were used to create a digital model of the riverbed morphology from which hillshade images and contours were generated. The survey was conducted from RM 0.0 (the southern end of Harbor Island) to the bridge at RM 4.8 (Map 2-3). The procedures used to conduct the survey were described in detail in the quality assurance project plan (QAPP) (Windward 2003d) and are briefly summarized below.

A boat equipped with Reson SeaBat® 8101 multibeam bathymetric sonar ran multiple lines parallel to the shoreline to achieve full bathymetric coverage. Several perpendicular cross-tie lines were also surveyed to confirm system calibration and document accuracy. A preliminary coverage plot was generated in real-time to show multibeam swath coverage.

Survey coverage met most of the requirements specified in the QAPP, with a few expected data gaps. Because of low clearance, the survey boat was not able to navigate under the bridge at S 102nd Street, which excluded the area from RM 4.8 to RM 5.0 from the survey. The survey was conducted in August, during a spring tide, so that it could be completed before the beginning of the Muckleshoot netfishing season. As a result, there were low tides during survey hours, even though shoreline surveying was timed so that data could be collected during the highest possible tide stage. The tidal cycle limited the ability of the survey crew to collect all shoreline data during a tide stage above the targeted 5 ft MLLW (as stated in the QAPP).

Other expected data gaps included areas where obstructions, such as docks, vessels, or pilings, restricted safe vessel operation. With the exception of these data gaps, the LDW multibeam bathymetric dataset was considered to provide a complete and accurate picture of the LDW bathymetry (Windward and DEA 2004).

2.3.2 LDW dredging history

As discussed in Section 2.2, the LDW has a long history of dredging, with events aimed at supporting general navigation needs as well as reworking the shoreline to accommodate new and updated infrastructure and berthing areas. USACE is responsible for continued maintenance dredging of the LDW navigation channel. Table 2-1 summarizes maintenance dredging events completed by USACE since 1986.

Table 2-1. LDW navigation channel maintenance dredging events since 1986

Location (RM)	Dredging Date		Volume Dredged (cy)	Paydepth/Overdepth (ft MLLW)[a]	Authorized Navigation Channel Depth (ft MLLW)	Dredging Distance		Reference
	Start	End				Station Start	Station End	
3.34 to 4.65	2/6/1992	3/21/1992	199,361	-15/-17	-15	205+00	275+56	USACE (1992)
3.43 to 4.65	3/11/1999	6/29/1999	165,116	-15/-16	-15	210+00	275+56	USACE (1999)
3.97 to 4.65	2/28/1990	3/30/1990	127,619	-17	-15	238+00	275+56	USACE (1990)
4.02 to 4.48	2/22/1996	3/30/1996	90,057	-15/-16	-15	241+00	265+00	USACE (1996)
4.19 to 4.38	3/11/1986	3/29/1986	33,637	-16/-18	-15	250+00	260+00	USACE (1986b)
4.26 to 4.65	2/5/1997	3/31/1997	89,011	-15/-16	-15	253+31	275+56	USACE (1997a)

Location (RM)	**Dredging Date**		**Volume Dredged (cy)**	**Paydepth/ Overdepth (ft MLLW)[a]**	**Authorized Navigation Channel Depth (ft MLLW)**	**Dredging Distance**		**Reference**
	Start	**End**				**Station Start**	**Station End**	
4.27 to 4.65	1/14/2002	2/9/2002	96,523	-15/-16	-15	254+00	275+56	USACE (2002a)
4.27 to 4.65	12/22/2007	1/10/2008	140,608	-15/-16	-15	254+00	275+56	Manson Construction (2008a)
4.33 to 4.65	3/7/1994	3/28/1994	57,243	-15/-17	-15	257+35	275+56	USACE (1994)
4.33 to 4.65	1/15/2004	2/16/2004	75,770	-15/-17	-15	257+00	275+56	USACE (2004)
4.38 to 4.65	6/19/1986	07/15/1986	126,470	-16/-18	-15	260+00	275+56	USACE (1986a)
4.38 to 4.65	2/24/1987	3/24/1987	80,160	-18/-20	-15	260+00	275+56	USACE (1987)

Note: Dredging information provided by AECOM (formerly ENSR).

[a] Paydepth is the planned design depth of the dredging event; overdepth is maximum allowed dredging depth.

cy – cubic yards
LDW – Lower Duwamish Waterway
MLLW – mean lower low water
RM – river mile
USACE – US Army Corps of Engineers

In addition, numerous dredging events have been completed in isolated areas of the LDW by private entities. Table 2-2 lists the private dredging events that have been conducted in the LDW since 1986. The approximate locations of these dredging events are shown on Map 2-4. For the most part, these were maintenance dredging events or were associated with infrastructure improvements. In addition, sediment remediation and capping events were recently completed in the vicinity of the Duwamish/Diagonal and Norfolk combined sewer overflow (CSO) and storm drain outfalls (in 2004 and 2003, respectively), and a thin layer of sand was placed next to the southwestern part of the Duwamish/Diagonal area in 2005. A sediment remediation effort was also completed adjacent to the south storm drain at the Boeing Developmental Center (BDC) in the vicinity of the Norfolk outfall in 2003.

Table 2-2. History of private dredging events in the LDW since 1986

Location	Project or Site Name	River Side	Year	Volume Dredged (cy)	Paydepth/ Overdepth (ft MLLW)[a]	Purpose	Suitable for Open Water Disposal?	Permit	Reference
RM 1.02 to RM 1.09	Lehigh Northwest	east	2004	9,000	-20/-21	maintenance dredging	DMMUs 1 and 3 (6,000 cy) suitable; DMMU 2 (3,000 cy) not suitable	USACE DAIS No. LEHIG-1-B-F-196	MCS (2004d), USACE et al. (2004)
RM 1.39 to RM 1.42	Duwamish Shipyard	west	multiple	unknown	-15 to -25/ unknown	maintain depth of basin behind dolphins	unknown	unknown	USACE (2002b)
RM 1.42 to RM 1.54	Glacier NW	west	2005	9,920	-34/authorized to -35	maintenance dredging and thin-layer cap	DMMU 1 (3,250 cy) suitable; DMMUs 2 and 3 (6,670 cy) not suitable	USACE Permit No. 92-2-00452 and USACE DAIS No. GLANW-1-B-F-183	PIE (2002), Anchor (2005), USACE et al. (2003)
RM 1.43 to RM 1.52	Lone Star Northwest – West Terminal	east	1986	unknown	unknown	maintenance dredging	not suitable (taken to upland site)	USACE Permit No. 92-2-00452	Hartman (1992), PIE (2002)
RM 1.43 to RM 1.52	Lone Star Northwest – West Terminal	west	1993	3,900	-35/-36	maintenance dredging	suitable	USACE Permit No. 92-2-00452	Hartman (1992), PIE (2002)
RM 1.56 to RM 1.75	James Hardie Gypsum	east	1999	10,000	-31/ unknown	maintenance dredging	4,540 cy (of 7,042 cy) suitable	Permit 95-2-00837	Spearman (1999)
RM 1.55 to RM 1.75	Lone Star – Hardie/Kaiser	east	1996	18,000	-30/-31	maintenance dredging and dock upgrade	DMMUs 1 to 3 (9,375 cy) not suitable; DMMUs 4 and 5 (8,625 cy) suitable	Permit 95-2-00837	Hartman (1995), USACE et al. (1995)
Slip 2	Glacier Ready Mix	east	2001	4,900	-15/-16	maintenance dredging	suitable	USACE Permit No. 2001-02-00528	USACE et al. (2001), PIE (2001)
RM 1.78 to RM 1.95 (two areas)	Terminal 115	west	1993	3,000	-15/ unknown	maintenance dredging, dolphin construction	suitable	Dredging Project No. 92-2-01363	USACE et al. (1993)
RM 2.39 to RM 2.49	Boyer	west	1998	8,000	-10/unknown	maintenance dredging	DMMUs 5 and 6 suitable; DMMUs 1 to 4 at Hurlen site	USACE Permit No. 98-2-00477	Hart Crowser (1998), WR Consulting (2004)

Table 2-2, cont. History of private dredging events in the LDW since 1986

LOCATION	PROJECT OR SITE NAME	RIVER SIDE	YEAR	VOLUME DREDGED (cy)	PAYDEPTH/ OVERDEPTH (ft MLLW)[a]	PURPOSE	SUITABLE FOR OPEN WATER DISPOSAL?	PERMIT	REFERENCE
RM 2.45 to RM 2.47	Boyer	west	2004	unknown	unknown	dock replacement	unknown	USACE Nationwide Permit No. 3, Reference No. 200200607	WR Consulting (2004)
RM 2.64 to RM 2.77	Hurlen	west	1998	15,000	-10/unknown	maintenance dredging	DMMUs 1 and 4 suitable; DMMUs 2 and 3 not suitable	USACE Permit No. 98-2-00476	Hart Crowser (1998)
RM 2.79 to RM 2.85	Crowley	Slip 4	1996	13,000	-15/unknown	maintenance dredging	DMMU 2 (3,250 cy) suitable; DMMUs 1, 3, and 4 (9,750 cy) not suitable	USACE Permit OYB 95-2-00537	PTI (1995a, b), USACE et al. (1996)
RM 2.86 to RM 2.97	Morton	west	1992	7,980	-18/unknown	maintenance dredging	suitable	OYB-2-013054, City of Seattle Shoreline Permit No. 8903261-1991	Spearman (1991a), USACE et al. (1991)
RM 3.36 to RM 3.44	South Park Marina	west	1993	15,500	-8/-9	maintenance dredging	8,000 cy suitable	USACE Permit OYB-2-012574	Spearman (1991b; 2003)
RM 4.03 to RM 4.15	Duwamish Yacht Club	west	1999	24,000	-8/unknown	maintenance dredging	suitable	071-OYB-2-008104 and 071-OYB-2-012184	Hart Crowser (1999), USACE (1999)
RM 4.17 to RM 4.24	Delta Marine	west	2004	7,000	-10/-11 in one area; -15/-17 in another area	maintenance dredging	suitable	USACE Permit No. 2002-2-00175; DAIS No. DELTA-1-B-F-171.	USACE et al. (2002)
RM 4.17 to RM 4.24	Delta Marine	west	2008	3,550	-10/-11 to -15/-17, three areas	maintenance dredging	11,905 cy suitable	USACE Permit 200200175; DE Permit NWS-2008-0320-NO, DAIS No. DELTA-1-A-F-244	King County (2007a); Manson Construction (2008b);USACE et al. (2007); USACE (2008); USACE and Ecology (2008)

Note: Dredging information provided by AECOM (formerly ENSR).

[a] Paydepth is the planned design depth of the dredging event; overdepth is the maximum allowed dredging depth.

cy – cubic yards
DAIS – Dredge Analysis Information System
DMMO – Dredged Material Management Office
DMMU – dredge material management unit
MLLW – mean lower low water
PIE – Pacific International Engineering, PLLC
RM – river mile
SEPA – State Environmental Policy Act
USACE – US Army Corps of Engineers

2.4 METEOROLOGY

The climate of the Puget Sound area is characterized as "Pacific marine." The prevailing winds move moist air inland from the Pacific Ocean, moderating both winter and summer temperatures. According to the Western Regional Climate Center (WRCC), which has compiled meteorological data from 1931 to 2007 (WRCC 2007), monthly average winter temperatures at Sea-Tac International Airport (December through February) range from 4 to 6°C (40 to 43°F), and monthly average summer temperatures (June through August) range from 16 to 18°C (60 to 65°F). Over 50% of the annual precipitation in the region falls between the months of October and January. Annual monthly average precipitation at Sea-Tac International Airport ranges from 2 to 15 cm (0.8 to 6.0 in.), with an annual average precipitation of 97 cm (38 in.) (WRCC 2007). Winds in the Duwamish Valley are typically from the south and southwest in the late fall, winter, and spring and from the north in the summer and early fall (PSCAA 2007c; Battelle et al. 2001). Annual monthly average wind speeds at Sea-Tac International Airport range from 11 to 14 km/hr (7 to 9 mi/hr) (WRCC 2007).

2.5 HYDROGEOLOGY

The hydrogeology of the Duwamish River basin, including the nature of subsurface materials and geologic units, is complex and influences both surface water and groundwater flow systems within the basin. Regional recharge and discharge patterns also play a significant role in defining the groundwater flow system of the Duwamish basin. The following subsections provide an overview of the relevant geologic features that influence the groundwater flow pathways to the LDW.

2.5.1 Geologic history

The Greater Duwamish Valley was formed by the carving action of glaciers that last advanced into this area from British Columbia approximately 15,000 years ago. When the ice sheets began to retreat approximately 5,700 years ago, the waters of Puget Sound extended up the Duwamish Valley as far south as Auburn, about 32 km (19 mi) upstream of the present mouth of the LDW at Elliott Bay. Around that same time, the Osceola Mudflow descended from Mount Rainier, depositing a massive layer of sediment into the then marine waters near present-day Auburn and Kent. The mudflow diverted the historical course of the White River, at that time a tributary of the Puyallup River, to the Green River (Booth and Herman 1998).

The alluvial fill within the Duwamish Valley deepened over time from the deposition of upstream fluvial sediments of the White, Green, and Black Rivers, advancing the mouth of the Duwamish River farther to the north. The fill included beds of fine silts and sands deposited as riverine and floodplain deposits, with coarser sands and gravels deposited near the water's edge. These sediments eventually buried the post-glacial form of the valley so that only a few outcroppings of bedrock remain exposed at the ground surface. As the river flooded and migrated back and forth across the floodplain, these sediments were redeposited by the river and continually intermixed with additional riverine and floodplain deposits (Booth and Herman 1998).

In the late 1800s and early 1900s, the river was extensively modified. Tide flats and floodplains were filled to straighten the river channel, resulting in the abandonment of almost 6 km (3.7 mi) of the original meandering river bed. Current side slips in the LDW are remnants of these old river meanders (Map 2-1).

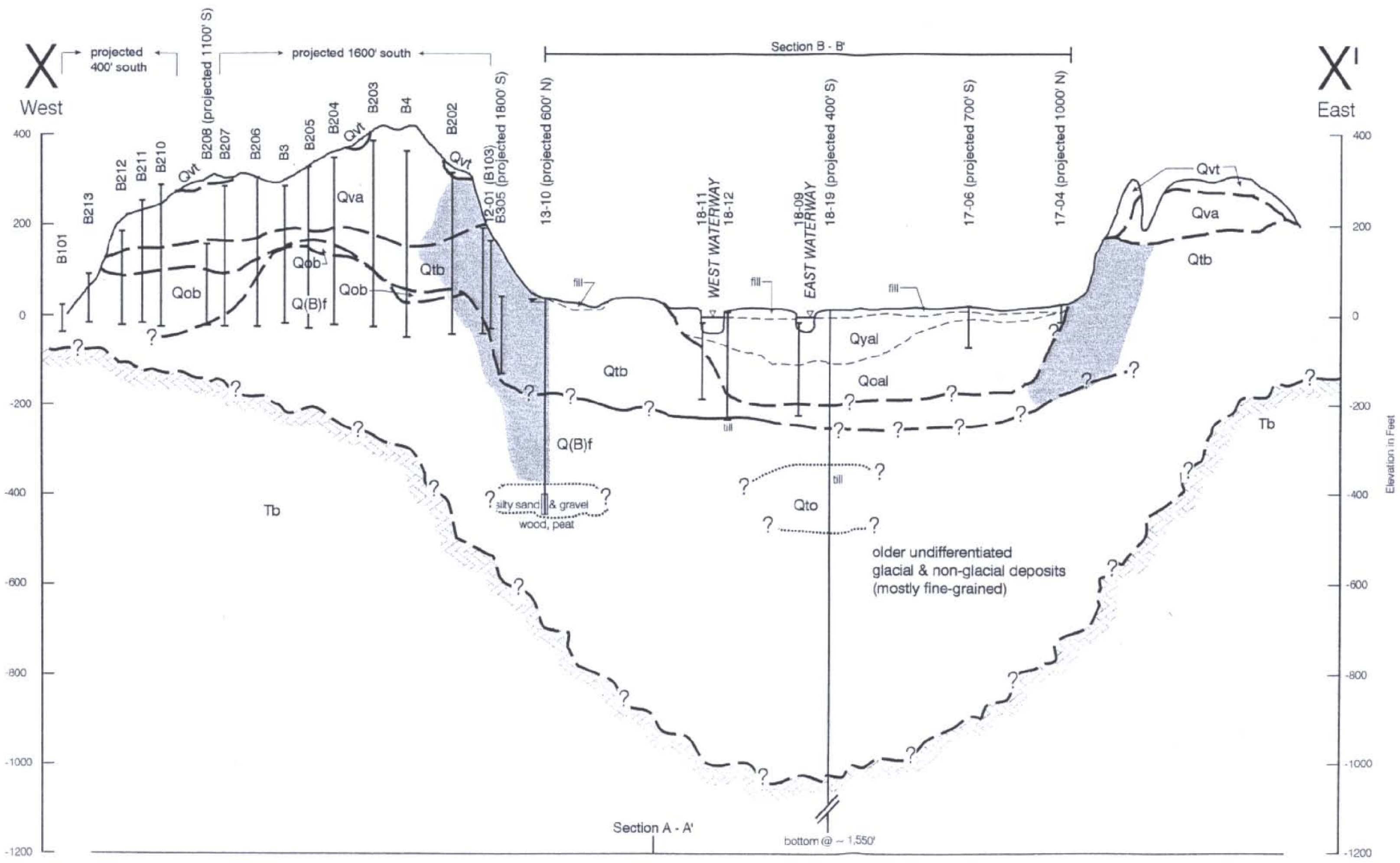
The channel has been frequently dredged for navigational purposes, and the excavated material was used to fill the old channel areas and the lowlands to bring them above flood levels. Subsequent filling of the lowlands for continued development resulted in a surficial layer of fill over most of the lower Duwamish Valley. Although the sediment types encountered in the LDW are variable (either from changing regional or local hydrodynamics or anthropogenic disturbances), basic sedimentary patterns of interbedded silts and sands are evident in the LDW. These patterns are controlled by Duwamish drainage basin morphology, precipitation, and flow rates. Both the regional and local aspects of these patterns are discussed in the following subsections.

2.5.2 Regional stratigraphy

The three principal geologic assemblages within the Greater Duwamish Valley that establish the hydrogeologic system, from oldest to youngest, are:

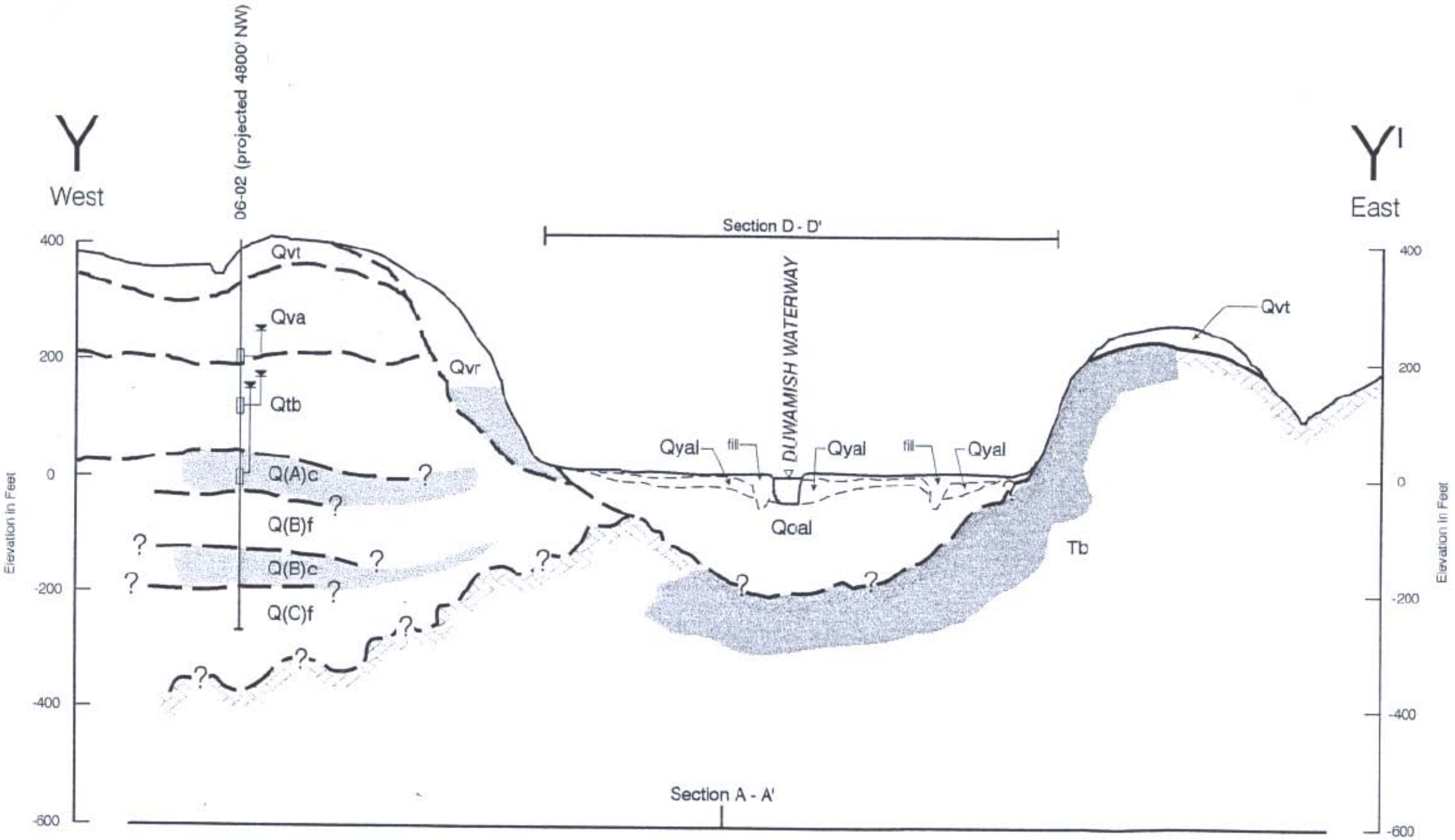
- Bedrock
- Glacial and non-glacial sedimentary units (glacially overridden and dense units that make up the plateaus to the east and west of the Duwamish Valley)
- Undifferentiated quaternary alluvial deposits (principal aquifer and groundwater pathway for the Duwamish basin)

Map 2-5 presents the surficial geology of the Greater Duwamish Valley basin and provides definitions of the surficial geologic units. Figures 2-2 and 2-3 are schematic cross sections of the regional stratigraphy at two locations in the Duwamish Valley as identified on Map 2-5 (cross sections X-X′ and Y-Y′, respectively); the additional cross sections identified on Map 2-5 are not included or discussed in this document. Figure 2-2 is a cross section of the LDW near Harbor Island, and Figure 2-3 presents the regional stratigraphy at approximately RM 3.0. Table 2-3 provides subsurface geologic unit nomenclature used in Figures 2-2 and 2-3 and discussed in this text (Booth and Herman 1998; 2005). The following subsections describe the assemblages and their influence on groundwater flow patterns in the Greater Duwamish Valley.



Source: Booth and Herman (1998)

Figure 2-2. Cross section of Greater Duwamish Valley stratigraphy at southern end of Harbor Island



Source: Booth and Herman (1998)

Figure 2-3. Cross section of Greater Duwamish Valley stratigraphy near LDW RM 3.0

Table 2-3. Definitions of subsurface geologic nomenclature

GEOLOGIC NOMENCLATURE	DEFINITION
Q(A)c	coarse-grained deposits, presumably immediately underlying Olympia-age (approximately 17,000 to 40,000 years ago) deposits (correlation is speculative)
Q(B)c	coarse-grained deposits, immediately underlying unit Q(B)f
Q(B)f	fine-grained deposits, immediately underlying the Q(A)c unit; speculative correlation would imply an age of at least 100,000 years old
Q(C)f	older undifferentiated, unconsolidated fine-grained deposits
Qoal	older alluvium
Qob	Olympia beds
Qtb	transitional beds, formed as the ice first advanced and laminated silt and clay were deposited; unit is a regionally significant aquitard, allowing very little movement of groundwater
Qva	coarser outwash, deposited by streams derived from the advancing ice sheet, initiated by the Vashon Advance Outwash (approximately 15,000 years ago); primary shallow aquifer in the upland areas throughout the region
Qvt	lodgment till, deposited by the melt-out of debris at the base of the glacier; composed of heterogeneous, compact sediment (glacial deposit)
Qyal	younger alluvium
Tb	the Blakely Formation, which includes marine sandstone, some conglomerate, and minor amounts of siltstone (bedrock)

Source: Booth and Herman (1998) and USGS (2005)

2.5.2.1 Bedrock

Bedrock in the Greater Duwamish Valley provides the lower boundary of the aquifer system and limits groundwater flow in the basin. At the northern end of the Duwamish Valley, the elevation of the bedrock unit ranges from roughly 60 m (200 ft) to over 500 m (1,640 ft) below ground surface (bgs). Exposed bedrock in the eastern and southern areas of the Greater Duwamish Valley (see Figures 2-2 and 2-3) is predominantly marine and continental sedimentary rocks intermixed with isolated areas of igneous rock deposited during the Tertiary period. Sedimentary rock units within the Greater Duwamish Valley are not an important source of groundwater because the predominantly cemented, fine-grained nature of the material precludes rapid groundwater movement. However, igneous rock layers are extensive in the area and can store and move water much more readily (Booth and Herman 1998).

2.5.2.2 Glacial and non-glacial sedimentary deposits

The glacial and non-glacial sedimentary units within the Duwamish basin are complex sequences of interbedded and unconsolidated deposits. In areas where bedrock occurs at significant depths below the river valley, these glacial sedimentary deposits serve as the lower boundary of the alluvial deposits in the Greater Duwamish Valley. The upland plateau areas to the east and west of the valley are formed predominantly of these glacially deposited sedimentary units (see Figures 2-2 and 2-3).

Little information is available on the glacially overridden sedimentary units within the LDW study area. These overridden deposits are mainly fine-grained materials; their maximum depth is unknown (Booth and Herman 1998). Although these deposits provide a geologic boundary to the alluvial deposits, they also provide a potential hydraulic pathway for the flow of upland groundwater to the Duwamish Valley alluvial sediments.

Thick sequences or silt beds (transitional beds) could potentially limit the upland inflow of groundwater where these deposits occur. The presence of saline water in the deeper alluvial sediments outside of current tidal influence areas suggests that there is little influx of fresh water into the original marine delta deposits. The lack of fresh groundwater in these deep alluvial sediments may indicate that the inflow of upland groundwater in this layer is limited (Figures 2-2 and 2-3).

2.5.2.3 Duwamish Valley alluvial deposits

The near-surface alluvial deposits in the Duwamish Valley extend to depths of roughly 60 m (200 ft) bgs within a trough bounded and underlain by either the bedrock unit or the dense glacial deposits and non-glacial sedimentary deposits. The geologic history of this valley suggests that the alluvial deposit sequences include estuarine deposits, typically fine sands and silts (often including shell fragments), which progress upward into more complex, interbedded river-dominated sequences of sand, silt, and gravel. These layers of alluvial deposits delineate the areas of advancing river delta sedimentation that increase in thickness from south to north (Booth and Herman 1998). Channel fill materials, which were deposited during settlement and when the river was straightened, are included in this category because fill material has properties similar to alluvium, as described below.

On a regional scale, the fill and alluvial deposits can be separated into various generalized units as illustrated in Figures 2-2 and 2-3. Geologic unit definitions and nomenclature are provided in Table 2-3. These regional units (from younger to older) include:

- **Fill –** The lower Duwamish River was straightened in the early 1900s into a navigation channel, using fill materials derived mostly from local sources. Much of the fill placed in the old river channels was dredged material, which is similar in hydraulic conductivity to the native younger alluvium. In the vicinity of the LDW, various depths of fill are present, ranging in thickness from 1 to 6 m (3 to 20 ft). Locally, the shallowest aquifer occurs within the lower portion of this fill material, especially in the northern sections of the LDW where upland was created during the last century. The depth of fill varied greatly and generally consisted of sand and silty sand in the saturated zone.

- **Younger alluvium (Qyal) –** This younger alluvium has been identified at the bottom of filled Duwamish River channels. Younger alluvium deposits are composed predominantly of sand, silt, gravel, and cobbles deposited by streams and running water (USGS 2005). In the central Duwamish Valley, roughly between RM 2.0 and RM 5.2, younger alluvial deposits are of relatively constant thickness and depth, based generally within 1.5 to 3 m (5 to 10 ft) of present-day mean sea level. These deposits are thicker in the south than in the north within the LDW; the thickest deposits on the northern end are estimated to occur at a depth of roughly 30 m (100 ft) bgs. The younger alluvium consists of sands, silts, and clays and includes abundant natural organic material. This layer is often distinguished from the overlying fill by abundant fibrous organic material typical of tide marsh deposits.
- **Older alluvium (Qoal**) – The older alluvium is characterized by estuarine deposits, often including shells at its lower depths, and composed of silts and clays with sandy interbeds (USGS 2005). This unit is commonly identified at depth between 15 and 30 m (50 and 100 ft) bgs in the central Duwamish Valley, increasing in depth toward the mouth of the LDW to between 45 and 60 m (150 and 200 ft) bgs. This unit has been best characterized at the Boeing properties in the central valley, where the older alluvium becomes finer-grained with increasing depth. In this area, the upper two-thirds of the older alluvium typically consist of sand and silty sand, and the lower third consists of sandy silt. The older alluvium also becomes significantly finer toward the north, with the sand almost completely absent near the mouth of the LDW. At this location, the layer is composed almost entirely of silt and clay, representing the farthest extent of the delta deposits into the marine waters and displaying the finest-grained material of the Duwamish Valley alluvial sequence.

2.5.3 LDW sediment stratigraphy

This section provides an overview of the stratigraphy of sediments identified in the LDW. Stratigraphy is the identification of individual sediment layers or groups of layers differentiated above or below by unity of color, texture, or gross appearance (Krumbein and Sloss 1963). The core logs provided in the subsurface sediment data report (Windward and RETEC 2007) characterize the stratigraphy of the units identified in the cores collected from within the LDW during the 2006 study.

The core logs provide the following stratigraphic information:

- A stratigraphic profile based on recovered depths, using description categories consistent with American Society for Testing and Materials (ASTM) nomenclature
- Detailed LDW sediment descriptions and comments
- A combined lithologic and stratigraphic profile based on *in situ* depths

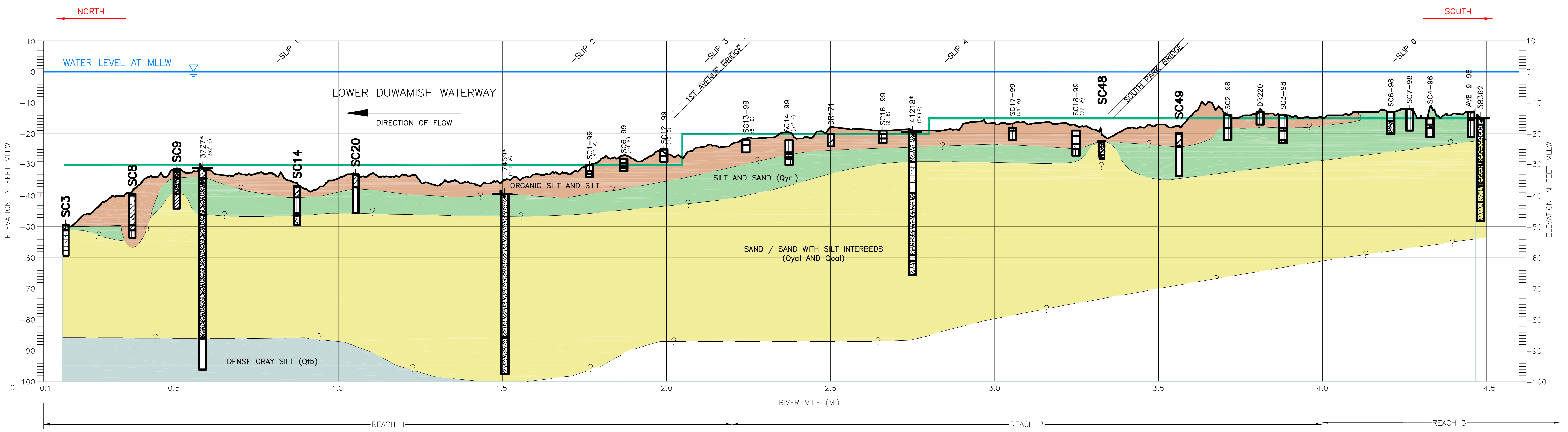
Based on field observations, the LDW sediments were grouped into three stratigraphic units. These units were delineated primarily based on density, color, sediment type, texture, and marker bed horizons; although other information, including the presence of anthropogenic matter, depth in the waterway, and available information on historical dredging events, was also considered.

Vertical sediment stratigraphic profiles were created for the LDW by comparing and evaluating sediment stratigraphy information from the LDW subsurface sampling event (Windward and RETEC 2007) and from other cores collected near the LDW shoreline. The core logs evaluated to prepare this presentation are provided in Appendix J. The locations of these cores are presented on maps included in Section 4.

The detailed stratigraphic information available from the cores collected in 2006, along with historical data from adjacent upland sites and other in-water boring data, was used to differentiate sedimentary units in the LDW from the larger undifferentiated Quaternary Alluvium unit of the Greater Duwamish Valley as described in Section 2.5.2.3.

To illustrate the distribution of these alluvial units in the LDW, a longitudinal cross section along the navigation channel centerline and three stratigraphic cross sections (A-A′, B-B′, and C-C′) at RM 1.0, RM 2.75, and RM 3.9 were prepared (Figures 2-4 through 2-7, respectively). The cross sections show the LDW at different river widths and together illustrate the change in sediment depositional patterns associated with the change in river morphology. The nomenclature used to describe the sediment units in these figures relates to the stratigraphy identified in the LDW and is not meant to correlate directly with the larger sedimentary units identified in the discussion of the Greater Duwamish Valley structure in Section 2.5.2.3.

File: L:\Lower Duwamish\X-SECTS 6-17-08\18220-LONG(c).dwg Layout: FIG 2-3 User: emarshall Plotted: Jun 17, 2008 - 4:24pm Xref's:



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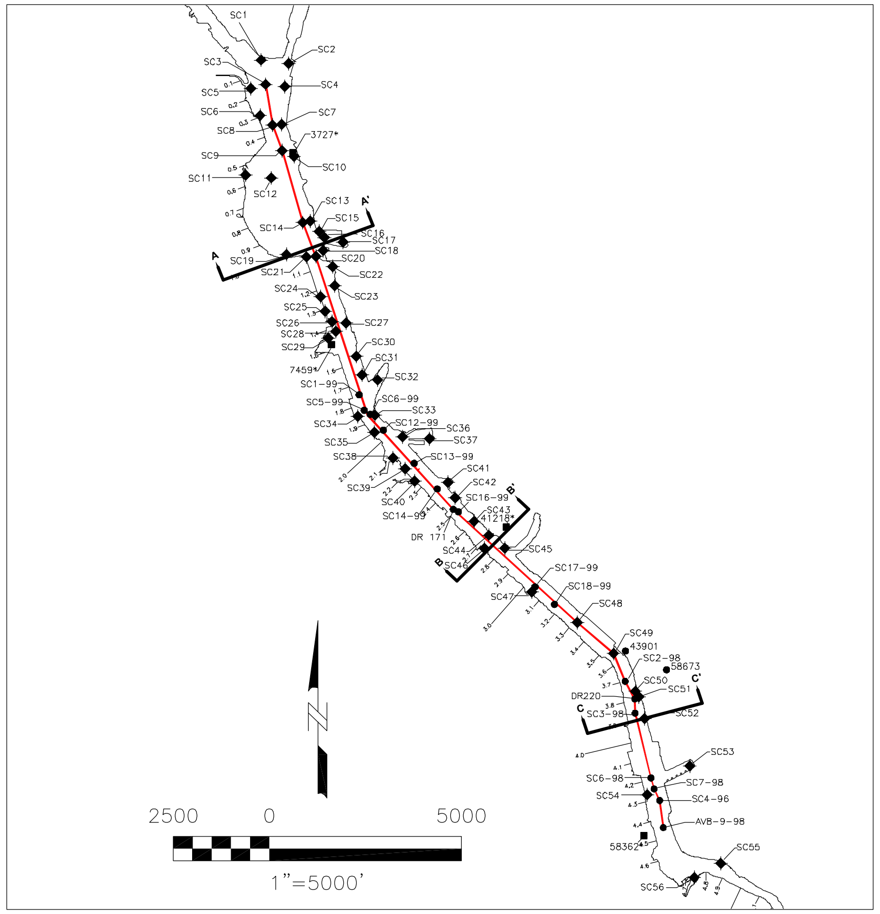
HORIZONTAL SCALE IN FEET
VERTICAL EXAGGERATION = 50:1



LEGEND

DR-171 ● Historical Subsurface Core
SC4-96 = PSSDA96 study
SC2-98 = PSDDA98 study
SC6-99 = PSDDA99 study
DR171 = EPASI study

3727* ■ Upland Core
7459 = Hart Crowser 1979 study
41218 = Yonemitsu Geological Services 1979 study
3727 = Dames and Moore 1988 study
58362 = Seattle Public Utilities (CPT-D77_01) 1985 study

SC48 ◆ 2006 Subsurface Core

Authorized Navigation Depth (ft MLLW)

Mid-channel mudline elevation (based on 2003 bathymetry survey, ft MLLW)

Water level at MLLW (ft)

KEY

SC14-99 (51' E) — Core Name (Distance from section to Core)
Top of Core
Observed Contacts
Bottom of Core

STRATIGRAPHY**

🟫 RECENT
🟩 UPPER ALLUVIUM/TRANSITION
🟨 LOWER ALLUVIUM
▨ DENSE POST-GLACIAL AND GLACIAL UNIT

Qtb = TRANSITIONAL SILTS
Qyal = YOUNGER ALLUVIUM
Qoal = OLDER ALLUVIUM

LITHOLOGY

- ▨ Clay/Silt
- ▨ Organic silt
- ☐ Silt
- ▩ Sand with silt
- ▤ Sand with silt interbeds
- ▩ Sand
- ▩ Gravel, sandy gravel

NOTES

1) MLLW – mean lower low water

2) 2006 Phase 2 cores (SC3, SC8, SC9, SC14, SC20, SC48, SC49) are labeled with bold font in cross section.

3) AV8-9-98 is an average of Core 8 and 9 from PSDDA (1998).

4) Upland cores from GeoMap NW, Pacific Northwest Center for Geologic Mapping Studies. http://geomapnw.ess.Washington.edu/index.php

5) Cross sections represent regional geology. See core logs in the subsurface sediment data report (Windward 2007) for detailed stratigraphy.

* Upland cores projected into Navigation Channel. Lower portions of upland cores used to bound deeper sediment units.

** Stratigraphy terminology is used in subsurface data report (Windward and Retec 2007) and sediment transport analysis report (Windward and QEA 2008).

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LOWER DUWAMISH WATERWAY
PHASE 2 RI REPORT
05482-015-514

NAVIGATIONAL CHANNEL LONGITUDINAL CROSS SECTION

DATE: 6/17/08
DRWN: E.M./SEA
FIGURE 2-4

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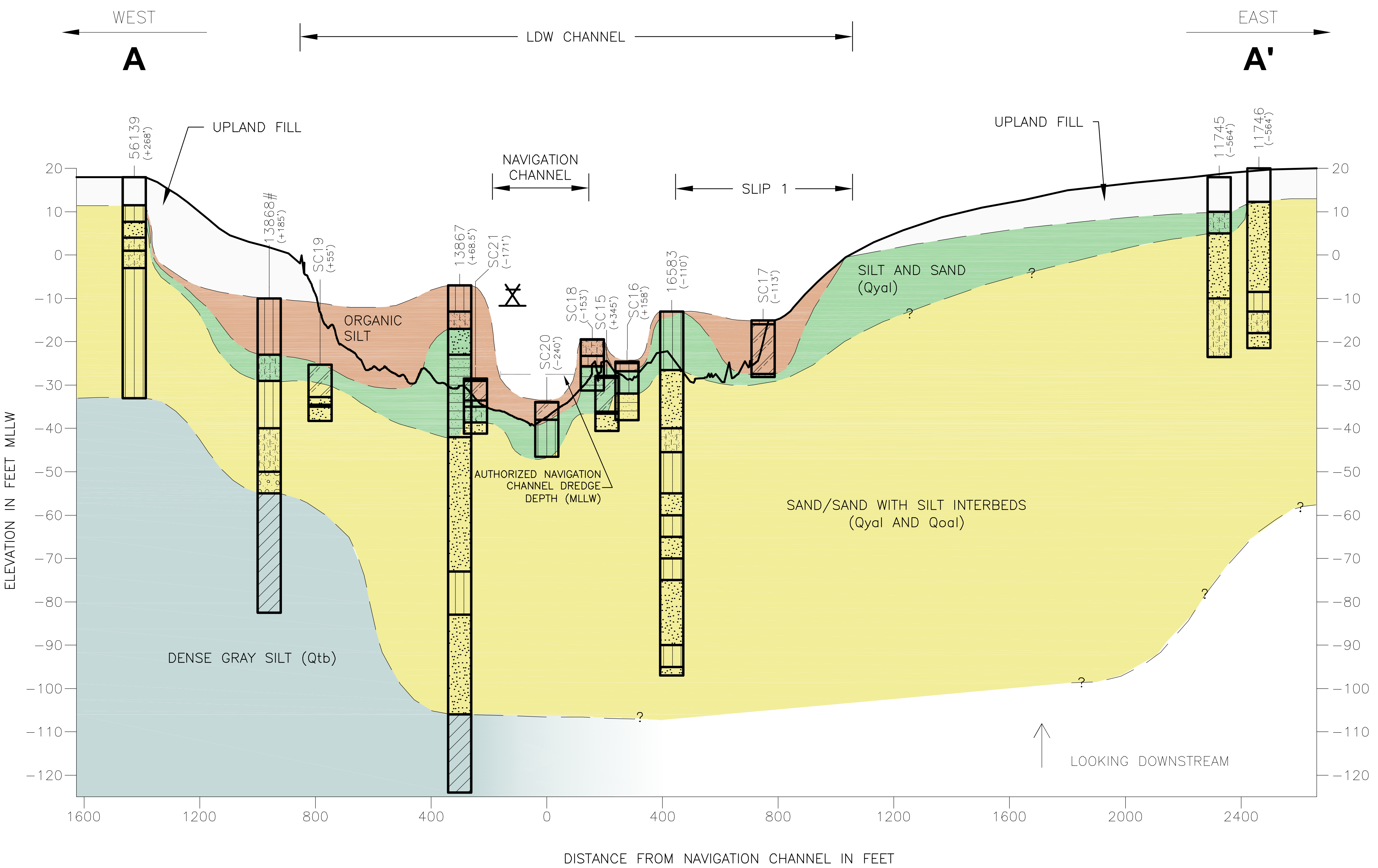

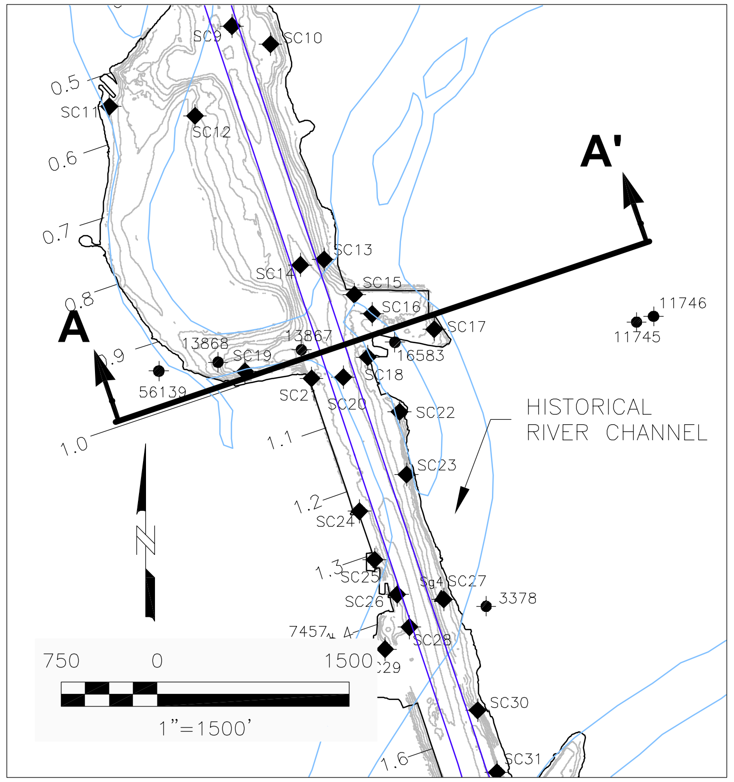

LITHOLOGY

- ☐ Fill
- ▨ Clay/silt
- ▨ Organic silt
- ▥ Silt
- ▦ Sand with silt
- ▤ Sand with silt interbeds
- ▩ Sand
- ☐ Gravel, sandy gravel

STRATIGRAPHY*

- ☐ UPLAND FILL
- 🟫 RECENT SEDIMENT
- 🟩 UPPER ALLUVIUM/TRANSITION
- 🟨 LOWER ALLUVIUM
- ⬜ DENSE POST-GLACIAL AND GLACIAL UNIT

LEGEND

INTERTIDAL OR SUBTIDAL CORES:

◆ SC52 – RI SUBSURFACE CORE (2006)

UPLAND BORINGS:

13867, 13868 –	TWELKER, 1970
11745, 11746 –	TERRA ASSOC., 1999
56139 –	GEORESOURCE INC., 1985
16583 –	GEOENGINEERS, 1995

Approximate mudline surface elevation in ft MLLW (2003 Bathymetry)

Approximate mudline surface elevation (1900 ACOE soundings)

KEY

SC19 (214') – Core Name – (Dist. from section to core in feet)

–Top of Core

–Observed Contacts

–Bottom of Core

NOTES:

1) Boring and core logs shown as in–situ depth. Actual contacts may vary.

2) Upland cores from GeoMap NW, Pacific Northwest Center for Geologic Mapping Studies. http://geomapnw.ess.Washington.edu/index.php

3) Cross sections represent regional geology. See core logs in the subsurface sediment data report (Windward 2007) for detailed stratigraphy.

* Stratigraphy terminology is used in subsurface data report (Windward 2007) and sediment transport analysis report (Windward and QEA 2007).

\# In water core (13868) projected onto upland portion of cross section. Top of core at –10ft MLLW. Mudline elevation line extended to top of upland core to west (56139).

200 0 400

SCALE IN FEET

VERTICAL EXAGGERATION = 15:1

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Lower Duwamish Waterway Group

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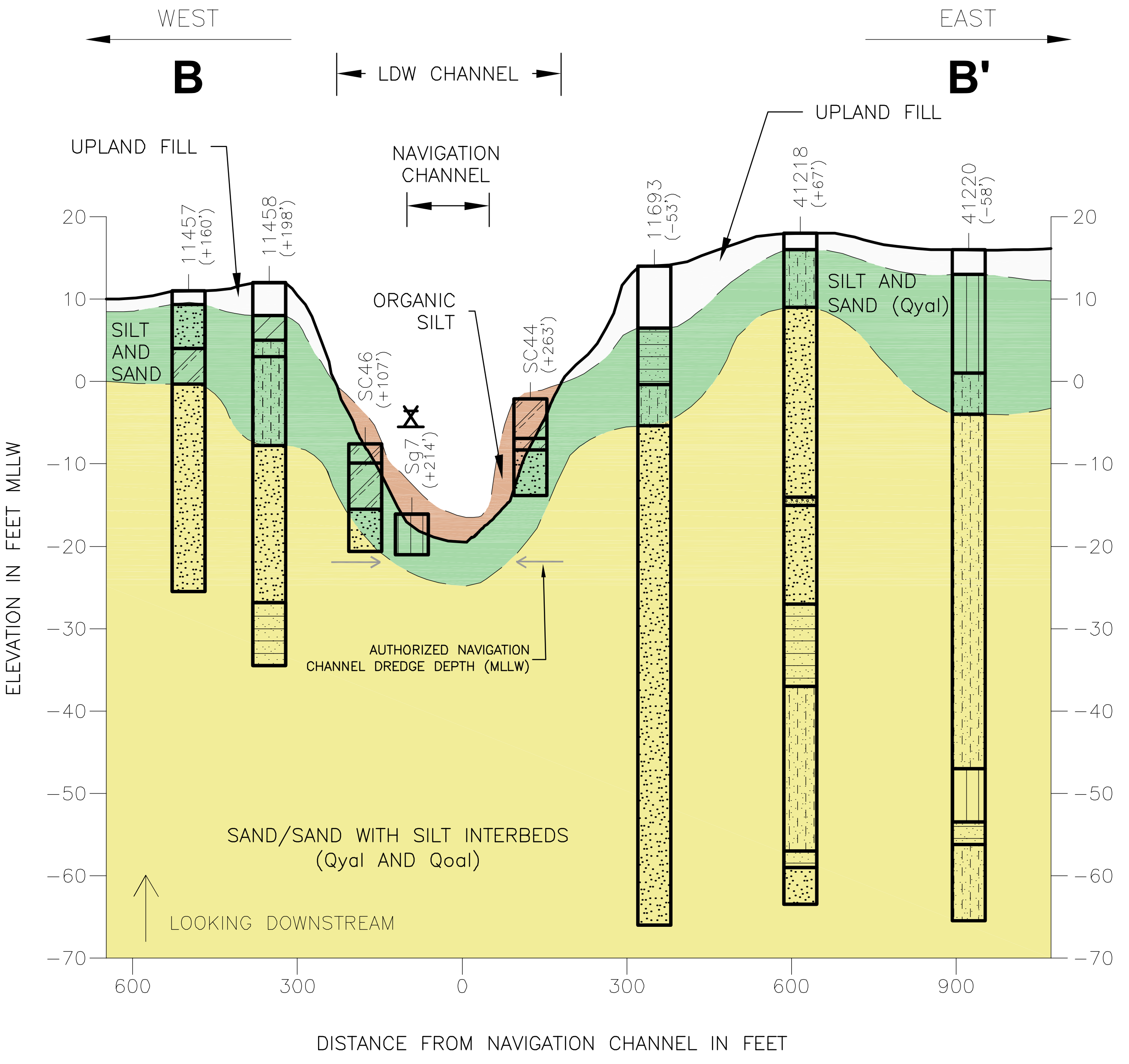
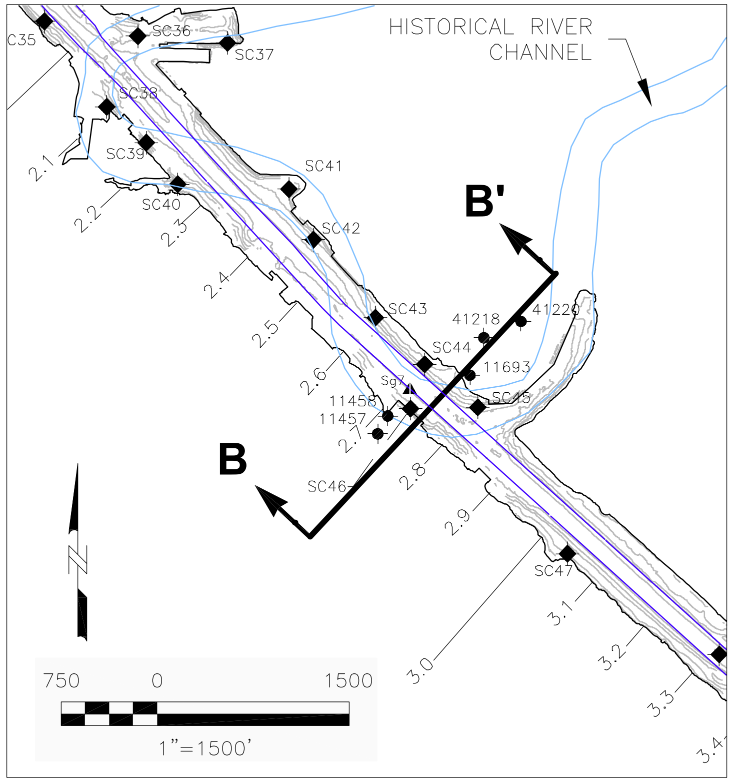
LOWER DUWAMISH WATERWAY
PHASE 2 RI
05482-015-514

DATE: 06/17/08 DRWN: E.M./SEA

GENERALIZED CROSS SECTION A-A'
RIVER MILE 1.0

FIGURE 2-5





NOTES:

1) Boring and core logs shown as in-situ depth. Actual contacts may vary.

2) Upland cores from GeoMap NW, Pacific Northwest Center for Geologic Mapping Studies. http://geomapnw.ess.Washington.edu/index.php

3) Cross sections represent regional geology. See core logs in the subsurface sediment data report (Windward 2007) for detailed stratigraphy.

* Stratigraphy terminology is used in subsurface data report (Windward 2007) and sediment transport analysis report (Windward and QEA 2007).

150 0 300

SCALE IN FEET

VERTICAL EXAGGERATION = 15:1

LITHOLOGY

- ☐ Fill
- ▨ Clay/silt
- ▨ Organic silt
- ▥ Silt
- ▦ Sand with silt
- ▤ Sand with silt interbeds
- ▩ Sand
- ▣ Gravel, sandy gravel

STRATIGRAPHY*

- ☐ UPLAND FILL
- 🟫 RECENT SEDIMENT
- 🟩 UPPER ALLUVIUM/TRANSITION
- 🟨 LOWER ALLUVIUM

LEGEND

INTERTIDAL OR SUBTIDAL CORES:

- ◆ SC44, SC46 – RI SUBSURFACE CORE (2006)
- ▲ Sg7 – GEOCHRONOLOGY CORE (2004)

UPLAND BORINGS:

- ● 11457, 11458 – CONVERSE CONSULTANTS 1988
- ● 11693 – GOLDER ASSOCIATES, 1997
- ● 41218, 41220 – YONEMITSU, 1979

Approximate mudline surface elevation in ft MLLW (2003 Bathymetry)

⌅ Approximate mudline surface elevation (1900 ACOE soundings)

KEY

SC19 (214') –Core Name –(Dist. from section to core in feet)

–Top of Core

–Observed Contacts

–Bottom of Core

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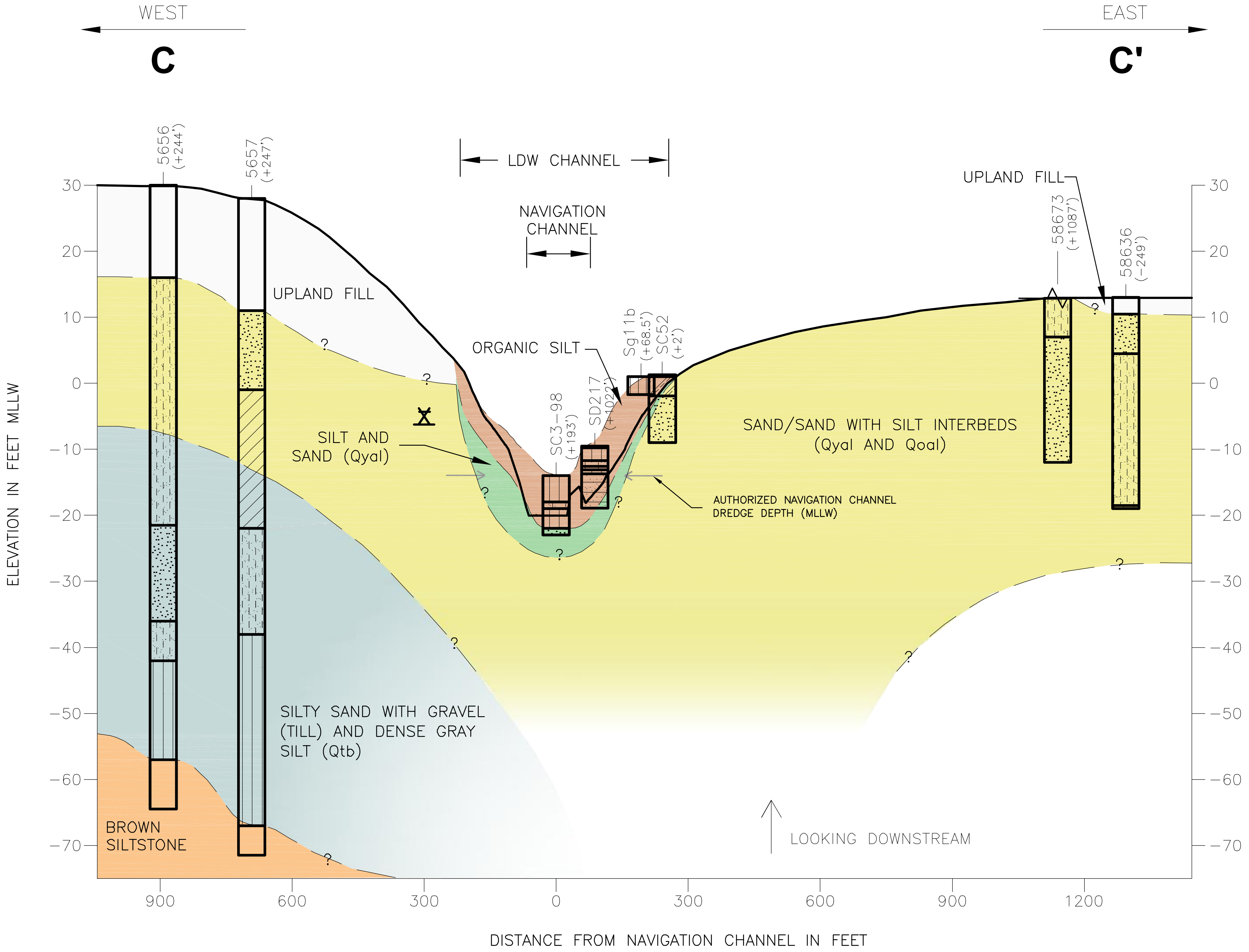
LOWER DUWAMISH WATERWAY
PHASE 2 RI
05482-015-514

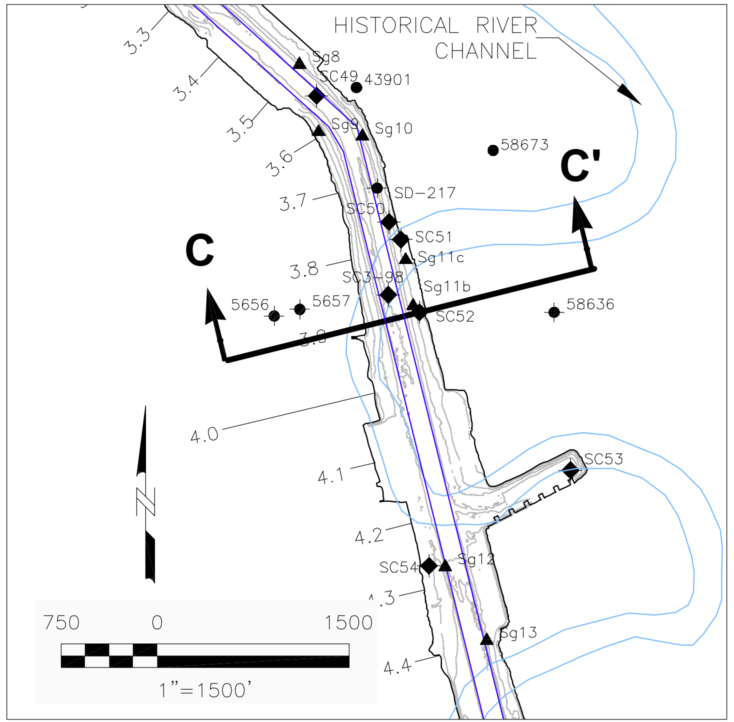
GENERALIZED CROSS SECTION B-B'
RIVER MILE 2.75

DATE: 06/17/08 | DRWN: E.M./SEA

FIGURE 2-6

File: L:\Lower Duwamish\X-SECTS 6-17-08\CROSS-SEC_B-B'.dwg Layout: fig 2-5 User: emarshall Plotted: Jun 17, 2008 - 4:20pm Xref's:





LEGEND

INTERTIDAL OR SUBTIDAL CORES:

◆ SC52 – RI SUBSURFACE CORE (2006)

▲ Sg11b – GEOCHRONOLOGY CORE (2004)

UPLAND BORINGS:

● 5656, 5657 – DAMES & MOORE, 1984

● 58636 – GEOENGINEERS, 1988

Approximate mudline surface elevation in ft MLLW (2003 Bathymetry)

Ẋ Approximate mudline surface elevation (1900 ACOE soundings)

LITHOLOGY

☐ Fill
▨ Clay/silt
▨ Organic silt
▥ Silt
▦ Sand with silt
▤ Sand with silt interbeds
▩ Sand
☐ Gravel, sandy gravel

STRATIGRAPHY*

☐ UPLAND FILL
🟫 RECENT SEDIMENT
🟩 UPPER ALLUVIUM/TRANSITION
🟨 LOWER ALLUVIUM
⬜ DENSE POST–GLACIAL AND GLACIAL UNIT
🟧 BEDROCK

KEY

SC19 (214') –Core Name –(Dist. from section to core in feet)
–Top of Core
–Observed Contacts
–Bottom of Core

150 0 300
SCALE IN FEET
VERTICAL EXAGGERATION = 15:1

NOTES:
1) Boring and core logs shown as in–situ depth. Actual contacts may vary.

2) Upland cores from GeoMap NW, Pacific Northwest Center for Geologic Mapping Studies. http://geomapnw.ess.Washington.edu/index.php

3) Cross sections represent regional geology. See core logs in the subsurface sediment data report (Windward 2007) for detailed stratigraphy.

* Stratigraphy terminology is used in subsurface data report (Windward 2007) and sediment transport analysis report (Windward and QEA 2007).

ENSR | AECOM

Lower Duwamish Waterway Group
Port of Seattle / City of Seattle / King County / The Boeing Company

LOWER DUWAMISH WATERWAY
PHASE 2 RI
05482-015-514

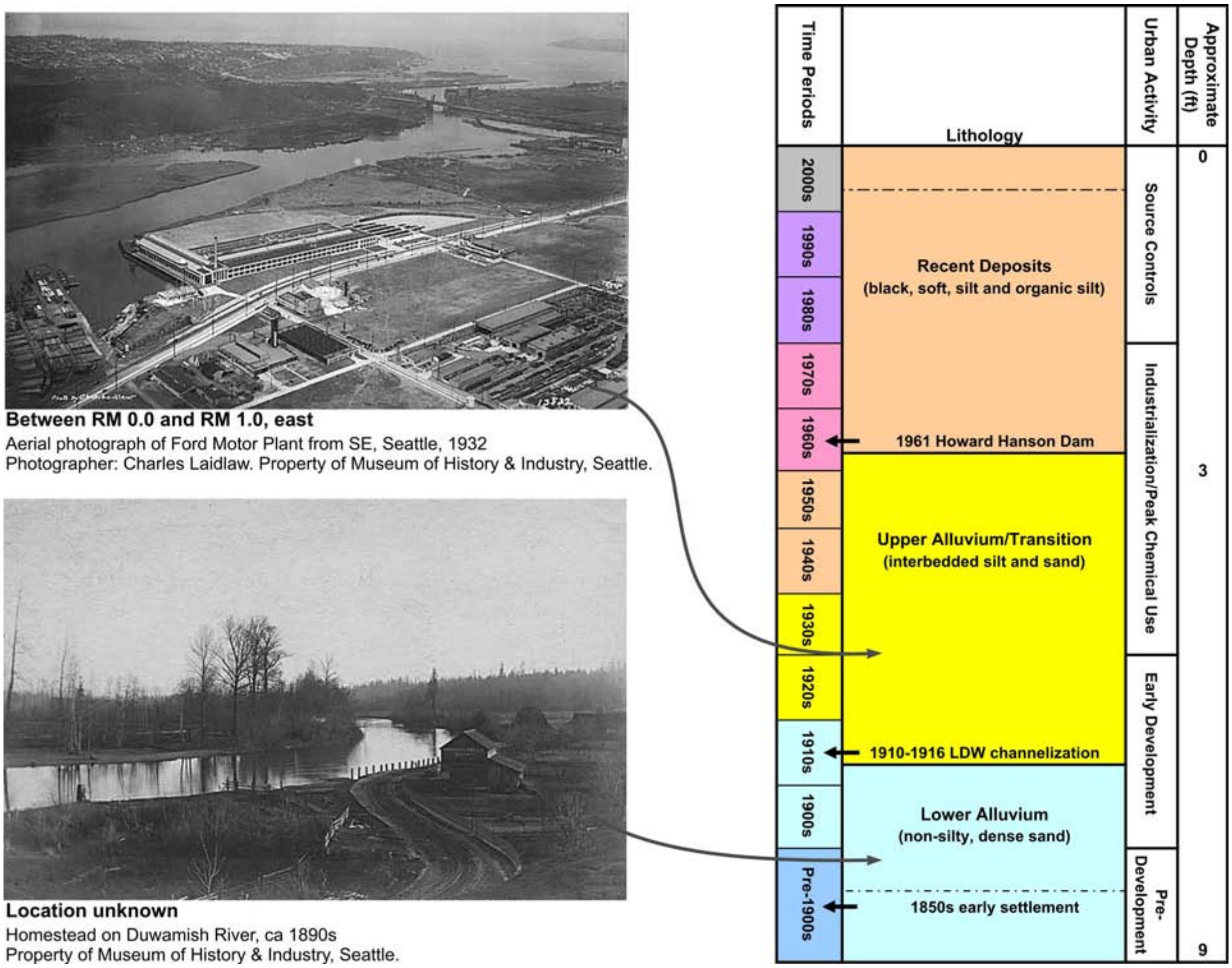
DATE: 06/17/08 DRWN: E.M./SEA

GENERALIZED CROSS SECTION C-C'
RIVER MILE 3.9

FIGURE 2-7

File: L:\Lower Duwamish\X-SECTS 6-17-08\CROSS-SEC_C-C'.dwg Layout: fig 2-6 User: emarshall Plotted: Jun 17, 2008 – 4:22pm Xref's:

Figure 2-8 provides a visual summary of sediment lithology in the LDW over the past 120 yrs relative to physical events and urban development.



Between RM 0.0 and RM 1.0, east
Aerial photograph of Ford Motor Plant from SE, Seattle, 1932
Photographer: Charles Laidlaw. Property of Museum of History & Industry, Seattle.

Location unknown
Homestead on Duwamish River, ca 1890s
Property of Museum of History & Industry, Seattle.

Figure courtesy of AECOM

Figure 2-8. **Summary of sediment lithology in the LDW relative to physical events and human development**

As shown in the cross sections (Figures 2-4 to 2-7), the LDW sediment typically consists of softer, recent sediments overlying transitional or upper alluvial sediments. In turn, these layers overlie a lower alluvial unit that represents deposition prior to the industrialization of the area.

In addition to the primary stratigraphic units presented in the cross sections, two other upland layers, fill and glacially–overridden sedimentary units, also have been identified in the study area. LDW sediments were classified as “other” when the sediment matrix was composed of over 50% fill, woody debris, or anthropogenic materials (Windward and RETEC 2007).

Three of the subsurface cores collected for the RI contained sediments classified as "other." Core SC2, located just north of RM 0.1 on the east side of the LDW, contained over 80% anthropogenic material between 4.3 and 10.5 ft bgs; the material was described as seams of silt and rock flour and chunky balls of hard silt (Windward and RETEC 2007). Traces of wood, a sheen, black streaks, and minor sand seams were also observed in core SC2 in the same sediment interval. Cores SC26 and SC28, located near one another at RM 1.4 on the west side of the LDW, also contained sediment intervals classified as "other." The sediment in the 7.7-to-9.1-ft (bgs) interval of core SC26 was described as "gravel with a hydrocarbon-like sheen and scattered debris, including possible paint chips and glass." Sediment from 5.8 to 12.8 ft bgs in core SC28 consisted of approximately 80% sand blast grit with a metallic shiny surface. Scattered debris, including possible paint chips and asphalt-like conglomerates, was associated with the sand blast grit in core SC28 (Windward and RETEC 2007).

2.5.4 Sediment lithology in the LDW

During the subsurface sediment investigations conducted for the RI, the lithology of the sediment cores collected was recorded in the field according to nomenclature established in ASTM D-2488, and a core log field key was used to ensure nomenclature consistency (Windward and RETEC 2007). Lithology is defined as the physical characteristics of the material described in terms of the dominant soil type (e.g., sand or silt), grain size percentages, texture (e.g., fine or medium grain), sorting (e.g., well sorted or poorly sorted), shape, and the color and mineralogy of particles (Krumbein and Sloss 1963).

Geotechnical parameter tests were also performed on a subset of subsurface sediment cores collected in 2006 within the upper 4 ft of the cores (Windward and RETEC 2007). These parameters included moisture content, specific gravity, Atterberg limits (i.e., liquid limit, plastic limit, plastic index), bulk density (dry and wet), and porosity.

Specific gravity, porosity, and wet density sample results did not vary notably with depth, indicating that sediment texture in the upper 4 ft was relatively uniform. The mean specific gravity of all subsurface sediment samples across similar core intervals ranged from 2.64 grams per cubic centimeter (g/cc) to 2.66 g/cc. The mean sediment porosity ranged from 0.59 standard unit (SU) to 0.64 SU, and the mean wet bulk density ranged from 102 pounds per cubic foot (pcf) to 104.4 pcf.

Other geotechnical properties varied with depth. The mean moisture content of all samples was 75% dry weight (dw) at the surface, decreasing to 63% dw below the 2-ft interval, consistent with the decrease in water content with depth as noted on the core logs. The mean dry bulk density across similar core intervals increased with depth from 60.4 to 67.2 pcf.

Atterberg limits testing was performed on fine-grained silt/clay sediments to determine the relationship between moisture content and LDW sediment type. The testing interprets the liquid limit of the sample (when sediment changes from a viscous fluid to a plastic state) and the plastic limit (where sediment behaves as a solid). The plasticity index is the difference between the liquid limit and the plastic limit. The mean liquid limit of all subsurface sediment samples ranged from 61.2% to 70.7% dw, and the mean plastic limit ranged from 35.0% to 39.3% dw. Sediment samples exhibited medium to high plasticity, with the mean plasticity index varying from 26.2% to 32.5% dw (Windward and RETEC 2007).

2.5.5 Sediment physical properties

Sediment composition varies greatly throughout the LDW, ranging from sands to mud (fine-grained silt and clays), depending on the source of the sediments and the local current velocity. LDW sediment typically consists of slightly sandy to sandy silt with varying amounts of organic material. Finer-grained sediments have typically been reported in remnant mudflats, along channel sideslopes, and within portions of the navigation channel. Sediments in the navigation channel near the Upper Turning Basin have been predominantly sands, whereas sediments toward the mouth of the river have been predominantly fine-grained silts.

Coarse sediments were present at the surface in nearshore areas adjacent to CSO and storm drain (SD) discharges (Weston 1999a). Only nine of the subsurface sediment cores collected as part of the RI contained intervals dominated by gravel; four of the cores were collected downstream of RM 0.6. The gravel interval was present in the upper alluvium/transition stratigraphic layer in three of those cores (i.e., near RM 0.1 on the east side and near RM 0.1 and RM 0.5 on the west side of the LDW) and in the recent stratigraphic layer of the fourth core (i.e., near RM 0.5 on the east side). Upstream of RM 0.6, gravel was identified in five additional cores. The gravel in one core collected from the west side of the LDW at RM 1.4 was associated with anthropogenic sources. Gravel was present in the upper alluvium/transition stratigraphic layer in two cores (i.e., near RM 1.4 and RM 4.7 on the west side) and in the recent stratigraphic layer in the other two cores (i.e., near Slip 2 and RM 2.2 west).

Overall, the percent fines (also referred to as silt+clay) content of surface sediment in the LDW has been reported to be highly variable, with an average content of 53% (Map 2-6). The average silt+clay content in the navigation channel was 62%; the 10th and 90th percentile silt+clay contents were 29 and 82%, respectively. Silt+clay content was more variable outside of the navigation channel (excluding the slips), with 10th and 90th percentile contents of 13 and 87%, respectively, and an average content of 53%. Average silt+clay contents have been calculated using point-based averages.

Three of the five slips along the LDW had high silt+clay contents relative to the overall LDW average. Slips 1, 3, and 6 had average silt+clay contents of 79, 71, and 87%, respectively. The silt+clay contents of Slips 2 and 4 were lower, with average values of 41% and 57%, respectively. The area above RM 5.0 had a relatively low average silt+clay content (approximately 11.5%).

Total organic carbon (TOC) content in surface sediment did not vary widely (Map 2-7). Outside the navigation channel, the 10th and 90th percentiles were 0.80 and 2.9%, respectively. The TOC content in the navigation channel was less variable than the TOC content outside the navigation channel, with 10th and 90th percentiles of 1.2 and 2.6%, respectively. The average TOC content (1.9%) was the same within and outside the navigation channel. The TOC content in Slips 1, 3, 4, and 6 was slightly higher than the LDW-wide average, with average TOC contents of 2.3, 2.2, 2.6, and 2.7%, respectively. In Slip 2, the average TOC content (1.5%) was lower than the LDW-wide average. Average TOC content was calculated using point-based averages. The areas upstream of RM 5.0 had a lower average TOC content (0.84%) and lower average silt+clay content (11.5%).

Woody debris has been observed in sediment throughout the LDW study area. A review of sediment characteristic data collected during the three rounds of surface sediment sampling conducted as part of the RI revealed that woody debris was observed in at least one surface sediment sample within almost every tenth of a river mile interval between RM 0.0 and RM 4.8. Woody debris was not observed in any of the six surface sediment samples collected within the LDW upstream of RM 4.9. Woody debris and wood fragments were also observed at various depth intervals in subsurface sediment cores collected throughout the LDW. One core, collected upstream of RM 4.8, had wood fragments within the 5-to-6-ft-interval below the mudline of this core. The prevalence of woody debris in the LDW is consistent with the waterway's history of being used for timber storage and transport.

2.5.6 Groundwater systems, transport, and processes

The previous sections characterized the geologic and stratigraphic composition of the soils and sediments in the Greater Duwamish Valley and the LDW study area. These geological features help define the framework for the aquifer systems and influence the flow and transport of groundwater within the Duwamish basin.

This section provides an overview of the regional and local groundwater systems within the Duwamish Valley, including the spatial flow patterns and characteristics of groundwater within the LDW study area. Various physical and biochemical processes that can affect the transport of chemicals in groundwater are also discussed in general. Section 9.4.6 provides more site-specific information on the groundwater characteristics and investigation efforts ongoing in the LDW.

2.5.6.1 Valley aquifer and groundwater flow system

Aquifer Characteristics in the Duwamish Valley

In general, groundwater within the Duwamish Valley is unconfined at depths up to 3 m (10 ft) bgs (Booth and Herman 1998). Regionally, the valley alluvium is a single, large aquifer system. The maximum depth of the alluvial aquifer in the study area extends to roughly 30 m (98 ft) bgs, although deeper areas are present in the central portion of the Duwamish Valley. The thickness of the alluvial aquifer lessens to the north of the valley, where finer-grained silts and clays dominate. The aquifer's thickness also lessens to between 10 and 12 m (33 and 39 ft) bgs toward the eastern and western margins of the valley (Booth and Herman 1998).

Site-specific studies in the LDW basin often subdivide the alluvial aquifer into shallow, intermediate, and deep zones. The shallow zone is generally located within the fill and/or younger alluvium (Qyal), and the deep zone is generally located within the older alluvium (Qoal). Shallow aquifer zones in the LDW are predominantly located in silty layers within interbedded sandier aquifer soils. In many areas, these shallow aquifers contain large amounts of organic material associated with the original river delta.

The separation of these shallow and deep zones is based partly on the presence of silt layers (aquitards), the presence of localized upward groundwater gradients, and the occurrence of pockets of saline/brackish groundwater. The silt layers are rarely continuous throughout the valley and often constrain local groundwater flow. The discontinuous nature of these silt layers allows for areas of hydraulic connection between the shallower and deeper alluvium zones.

General Characteristics of Groundwater Flow in the LDW Basin

The aquifer's flow characteristics vary depending on the nature of the materials that make up the local alluvium, the proximity to the river, and local tidal fluctuations. The elevation gradient between the glacially overridden deposits in the uplands and the LDW sediments creates a regional flow system with significant hydraulic potential for the transport of groundwater from the upland areas to the LDW. Groundwater elevations in the uplands were between 30 and 60 m (98 and 197 ft), while elevations in the valley aquifer were between 3 and 6 m (10 and 20 ft) relative to mean sea level. Appendix G of the Phase 1 RI (Windward 2003a) included a detailed groundwater pathways assessment for select properties along the LDW and presented groundwater flow patterns obtained from relevant site-specific investigations for these facilities. The groundwater elevation data presented in the appendix were for the shallowest aquifer system for which reliable data were available.

Few hydraulic conductivity data are available for the LDW; most of the available data are from investigations completed at properties on the east side of the waterway. Available data indicate that hydraulic conductivity varies greatly from site to site. Estimated hydraulic conductivity in the vicinity of the BDC ranged from 3.8×10^{-2} to 4.5×10^{-2} cm/sec, whereas the average hydraulic conductivity was 2.3×10^{-2} cm/sec at the Rhône-Poulenc facility to the north of the BDC (Windward 2003a). At the Kenworth Trucking/PACCAR facility, north of Rhône-Poulenc, hydraulic conductivity ranged from 1.5×10^{-3} to 4.9×10^{-3} cm/sec. Further north, on the Boeing Isaacson property, hydraulic conductivity ranged from 10×10^{-3} to 10×10^{-4} cm/sec.

Within the greater LDW basin, groundwater flow rates and gradients vary greatly. Examples of various site-specific horizontal gradients and flow rates are presented in Table 2-4.

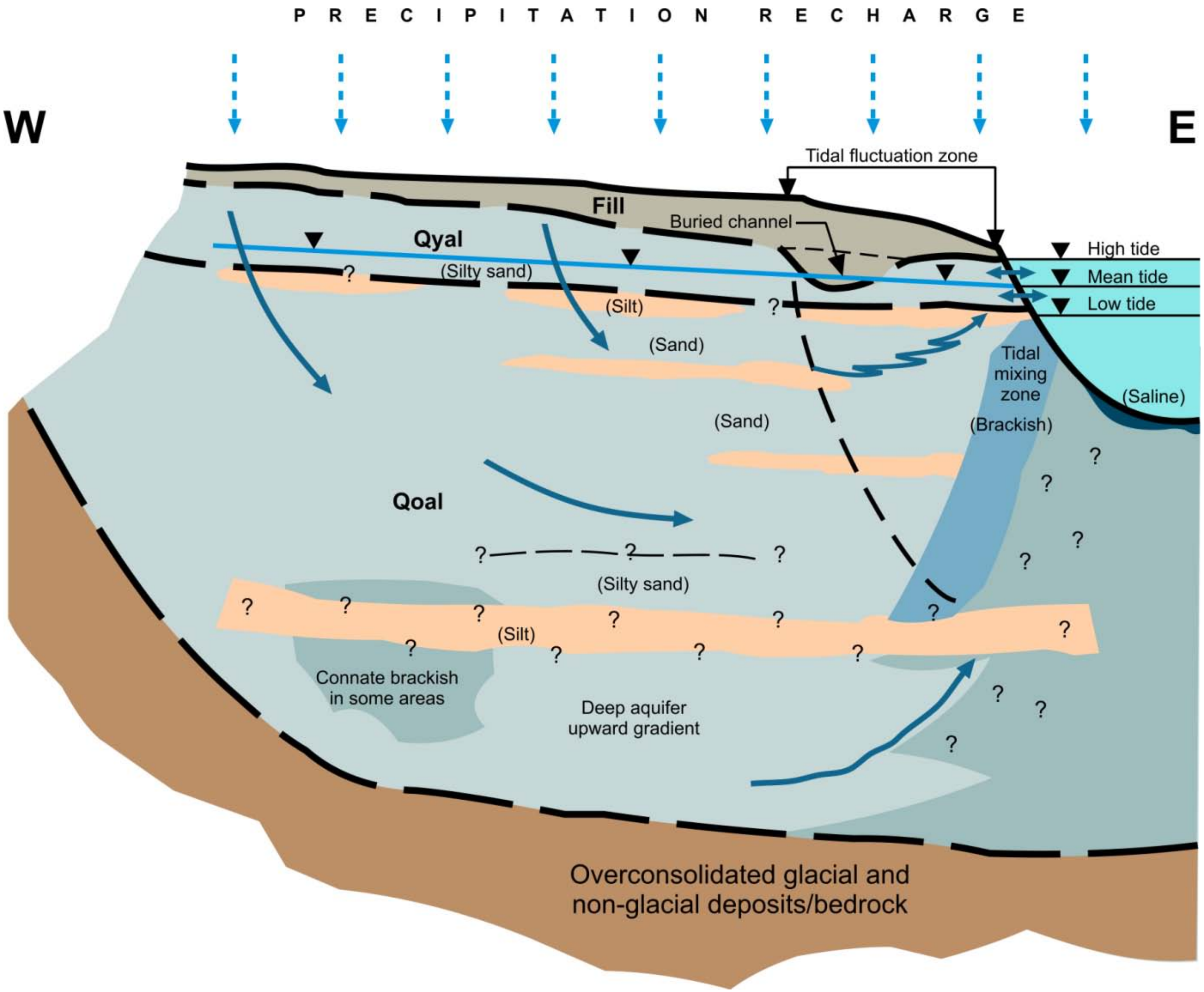
Table 2-4. Site-specific flow rates and gradients

LDW FACILITY	HORIZONTAL GRADIENTS (ft/ft)	FLOW RATES (ft/yr)
Boeing Developmental Center	0.0004 to 0.0008	65 to 150
Boeing Plant 2	0.0002 to 0.0029 (average 0.002)	25 to 26
Kenworth Trucking (PACCAR)	0.0017 (average)	12 to 24
Philips Services (Burlington Environmental)	0.0015 (average)	190
Rhône-Poulenc	0.002 (average)	25 to 30

LDW – Lower Duwamish Waterway

Groundwater flow within the LDW study area is generally toward the LDW. However, high tides in the LDW can cause temporary groundwater flow reversals. Tidal influences are greatest in wells installed in the sandiest alluvium in the central portion of the valley. Areas with tidal influence are generally within 100 to 150 m (328 to 492 ft) of the LDW shoreline (Booth and Herman 1998).

Tidal action can cause a continuous oscillation of groundwater within the effective zone of influence (Figure 2-9). At low tide, the hydraulic gradient between the groundwater system and the LDW is typically at its highest, causing the flow of local groundwater into the LDW. At high tide, the hydraulic gradient can reverse direction, causing LDW surface waters to flow into the adjacent LDW sediments and soil. The amount of LDW water intrusion into and out of the local aquifer depends on tidal patterns, site-specific aquifer conditions, soil stratigraphy (i.e., permeability), and rainfall recharge within the groundwater system.



Source: Modified from Aspect (2003)

Figure 2-9. Example of aquifer tidal mixing in the LDW

Recent tidal studies near the Boeing Isaacson property reported that tidal influences were noted in wells approximately 400 ft from the LDW; similar studies at the Boeing Plant 2 facility reported that tidal influences were noted between 300 and 600 ft from the LDW (Windward 2003a).

Tidal influence studies have also been conducted at Great Western International (GWI), located approximately 400 ft from the LDW shoreline. GWI has wells set in both the shallow and intermediate aquifer zones within the river valley. During high tide conditions, the flow gradient was toward the facility in both the upper and intermediate aquifer zones (Windward 2003a). During low-tide conditions, the flow gradient was toward the LDW (Table 2-5).

Table 2-5. Results of tidal influence study at GWI

Tide Condition	Aquifer Zone	Groundwater Flow Rate (ft/day)	
		Facility Wells	Tidal Zone Wells
High	shallow	0.80 (toward LDW)	0.93 (away from LDW)
	intermediate	0.75 (toward LDW)	0.25 (away from LDW)
Low	shallow	0.47 (toward LDW)	1.25 (toward LDW)
	intermediate	0.75 (toward LDW)	0.32 (toward LDW)

Source: Windward (2003a)
LDW – Lower Duwamish Waterway

Vertical Flow Gradients

Information on vertical flow gradients (both upward and downward) provided in the Duwamish groundwater pathways study (Booth and Herman 1998) indicated that downward flow gradients occur predominantly in the shallower alluvium layer, which extends roughly 10 to 15 m (33 to 49 ft) bgs, in part because of effects of local rainwater infiltration and groundwater recharge. The downward gradients are dependent on the volume of rainwater that infiltrates into the system (much of the LDW basin is classified as impervious), the presence of interbedded silt and sand layers (the greater the interbedding, the higher the downward gradient), and the distance from the waterway. Upward flow gradients are typically observed in groundwater wells installed in the deeper aquifer zone (Booth and Herman 1998).

Site-specific information, though limited, indicates that upward flow gradients have been identified in the vicinity of the Boeing Plant 2 facility, King County International Airport (KCIA), the South Park community, the BDC, and near Slip 6; these upward flow gradients were determined based on information from wells installed in the deeper aquifer zone (screen intervals between 40 and 80 ft bgs). Upward flow rates were lowest (between 0.002 and 0.07 m/day) beneath the eastern side of the Duwamish Valley (roughly between RM 2.0 and RM 5.2), where inflow from the adjacent uplands is limited by layers of bedrock and silt deposits. Higher upward flow rates, ranging from 0.02 to 0.3 m/day, generally occurred in the upland areas to the west of the LDW (Windward 2003a).

Where downward gradients intersect with upward gradients, the interaction has the potential to cause shallower groundwater to flow toward the LDW, often discharging as seeps in the intertidal zone (Figure 2-9). In addition, the presence of brackish and saline water tends to concentrate the outflow of the surficial aquifer into the intertidal areas because the less-dense, fresh groundwater tends to migrate above the higher-density saline water. This density variance minimizes the potential for shallower groundwater to mix with the more brackish or saline zones. Brackish and saline water areas have been primarily identified in the eastern portion of the Duwamish Valley as a result of extensive areas of impervious surface and limited freshwater recharge (Booth and Herman 1998).

2.5.6.2 Processes and conditions that can affect chemical transport in groundwater

The processes and conditions that can influence the movement of chemicals through the subsurface are complex. Determining whether a chemical in groundwater may or may not reach nearby surface water or sediment requires an analysis of site-specific conditions, flow patterns and rates, and chemical-specific attributes, such as the partitioning affinity and persistence of the chemical. The following sections provide an overview of the processes that could potentially affect the movement of chemicals in groundwater. Appendix G of the Phase 1 RI, as summarized in Section 9, provided a preliminary analysis of the potential for groundwater chemicals to reach the LDW at 12 sites judged by EPA and Ecology in 2002 to be the areas with the highest potential groundwater contamination. Site-specific seep, porewater, [13] and sediment testing in the vicinity of these areas was conducted and evaluated with respect to the potential for chemicals in groundwater to contaminate sediment (see Section 9.4.6).

Several processes can influence chemical concentrations in groundwater downgradient of a contaminated area, as well as concentrations in waters that discharge to the LDW. These processes include:

- **Advection** – Advective flow rates and patterns can transport dissolved chemicals and chemicals sorbed to small particles through groundwater to surface waters. Advection, the movement of groundwater through soil pores, is primarily dependent on the specific characteristics of the aquifer, such as porosity, hydraulic conductivity, and discontinuities within the aquifer system.
- **Partitioning** – The migration rate of dissolved chemicals in groundwater can be affected by chemical partitioning to the solid materials within the aquifer. The rate at which chemical transport is slowed (the retardation factor) is a function of the dry bulk density and porosity of the aquifer and the soil-water partitioning coefficient of the chemical (Fetter 1993). A soil-water partitioning coefficient for most organic compounds can be estimated as the product of the organic carbon (OC) content of the aquifer unit and the specific chemical's water-OC partitioning coefficient. For polar substances (e.g., pentachlorophenol), the water-OC coefficient is also dependent on groundwater pH, with the coefficient decreasing with increasing pH.
- **Volatilization –** The transition of a chemical from an aqueous phase into a vapor phase in unsaturated soil above the groundwater table that can occur when low-molecular-weight compounds migrate in groundwater.

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[13] Porewater is the interstitial water in sediment. Chemical concentrations in porewater reflect the state of chemical partitioning between sediment and interstitial water; the concentrations can also be affected by groundwater discharge if a porewater sample is collected from within a groundwater discharge zone.

- **Diffusion –** The movement of molecules along chemical concentration gradients; generally more significant at low groundwater velocities (e.g., through silt and clay layers).
- **Dispersion –** A small-scale mixing process that results from the various paths that individual chemical molecules take while traversing through porous soil or sediment. A substantial amount of dispersion can occur in fine silty sand sediment from constant tidal oscillation.
- **Dilution –** The decrease in concentration that occurs along the groundwater pathway from an area with elevated groundwater chemical concentrations as a result of recharge from precipitation. Dilution also occurs at the point of discharge into a tidally influenced water body as a result of water exchange between the surface water body and the discharging aquifer. Tidal monitoring studies have indicated that water intrudes from a surface water body into the adjacent aquifer during high tides and that groundwater discharges to surface water at low tides. This exchange can reduce the dissolved chemical concentrations discharging into the LDW (Yim and Mohsen 1992).
- **Biological degradation** – The degradation of non-chlorinated hydrocarbon compounds, including total petroleum hydrocarbons (TPH), low-molecular-weight polycyclic aromatic hydrocarbons (LPAHs), and benzene, toluene, ethylbenzene, and xylene (BTEX) compounds, under aerobic conditions (Wiedemeir et al. 1994). Hydrocarbon compounds can also be degraded by sulfate- and nitrate-reducing bacteria, although this process is slower than aerobic degradation (Wiedemeir et al. 1994). In general, organic compounds must come into contact with the cell membranes of micro-organisms to be degraded, so degradation rates are generally higher for more soluble compounds (Mackay et al. 1992). Because micro-organisms can also influence the reduction-oxidation potential in the groundwater environment (e.g., through microbial oxidation of organic matter) (Moore et al. 1988), they can also alter the speciation and mobility of metals. Reducing conditions can mobilize iron, manganese, arsenic, and other metals present in or sorbed onto sediment particles. Under aerobic conditions, the mobility of these metals is decreased as a result of sorption to or co-precipitation with iron and manganese (hydr)oxides (Moore et al. 1988). The combination of these processes can produce a transition zone between the deeper anaerobic sediment layers and the surface aerobic sediment layers.
- **Facilitated transport** – A process that can potentially accelerate the migration of a chemical beyond what would normally be expected. One example of facilitated transport involves co-solvent effects, in which high concentrations of organic or solvent compounds can mobilize hydrophobic compounds in groundwater. Colloidal processes can also facilitate transport whereby the

presence of large organic macromolecules can increase the mobility of hydrophobic chemicals (Huling 1989).

The net result of these processes on chemical transport is a function of numerous factors, including, but not limited to, the salinity, pH, reduction-oxidation potential, duration of contact between groundwater and contaminated soil, and the chemical properties of the contaminant.

Conditions within LDW groundwater are dependent on a range of factors, including (but not limited to) proximity to the LDW, depth, geology, tidal fluctuations, groundwater quality parameters (e.g., dissolved oxygen [DO], pH), and the specific attributes of chemicals that may be present in the groundwater. These various factors affect organic chemicals and metals in different ways.

Tidal intrusion during high-tide events can greatly influence nearshore groundwater quality. Tidal fluctuations in the LDW create a dynamic mixture of fresh and saline waters, which contain high concentrations of chloride, carbonate, and sulfide in downstream portions of the waterway. Furthermore, the infusion of LDW water into local groundwater can affect chemical mobility and persistence in a number of ways. For example, high DO levels can cause organic compounds (i.e., petroleum hydrocarbons) to biodegrade and can increase the extent to which dissolved metals precipitate out of solution or sorb to particles within the aquifer. Tidal intrusion can also dilute the concentrations of chemicals in groundwater prior to groundwater discharge into the LDW. Specific conductivity, often used to interpret levels of salinity in groundwater, was high (> 10,000 µmhos/cm) in the vicinity of the Boeing Plant 2 facility and KCIA (Booth and Herman 1998).

Groundwater pH can be affected by several factors, including the residence time or age of the groundwater (with older groundwater typically having higher pH), the mineralogy of the geologic materials surrounding the aquifer, the geochemistry of the aquifer system, and microbial activity (Ayotte et al. 2003; USGS 2007a). The pH of groundwater can be increased as a result of ion exchange in aquifers formerly inundated by seawater. Anthropogenic activities, including industrial operations, impacts from the use of underground storage tanks, and land disposal of wastes can also affect the groundwater pH. For example, the use of fertilizers has been found to reduce groundwater pH (Robinson 2003), and organic matter in groundwater can act as a pH buffer (Robinson 2003). The pH of groundwater can affect the solubility of metals and other ions and thus plays a role in groundwater quality.

Numerous groundwater studies that have been completed at facilities near the LDW show that pH and DO levels vary greatly by location and distance from the waterway (Table 2-6).

Table 2-6. Groundwater pH and dissolved oxygen data from a subset of the wells along the LDW

LDW Facility	Facility Location[a]	Well Location[b]	Aquifer Zone[c]	pH	Dissolved Oxygen (mg/L)
Terminal 108	RM 0.5, east side of LDW	site-wide (range)	shallow	5.92 – 7.10	0.36 – 2.32
		150 ft from LDW shoreline	shallow	6.14 – 7.10	0.47 – 1.73
		400 ft from LDW shoreline	shallow	6.03, 6.34	0.53, 1.36
		950 ft from LDW shoreline	shallow	6.36	2.32
Duwamish Shipyard	RM 1.3, west side of LDW	500 ft from LDW shoreline	shallow	6.45 – 6.49	1.00 – 1.19
Trotsky property	RM 2.2, west side of LDW	275 ft from LDW shoreline[d]	shallow	5.9	na
		475 ft from LDW shoreline[d]	shallow	8.0	na
Boeing Plant 2	RM 2.9 – RM 3.6, east side of LDW	shoreline wells (throughout property shoreline)	various depths	6.22 – 7.63	0.45 – 10.98[e]
		shoreline	shallow	6.30 – 7.56	0.45 – 10.98
		shoreline	intermediate	6.35 – 6.80	0.68 – 1.05
		shoreline	deep	6.22 – 7.63	0.47 – 5.12[e]
Long Painting	RM 3.0, west side of LDW	75 ft from LDW shoreline	shallow	6.06[f]	3.60[f]
Terminal 117	RM 3.6, west side of LDW	shoreline wells (throughout property shoreline)	shallow	6.12 – 8.35	0.38 – 9.9
		40 ft from LDW shoreline	shallow	5.84	4.52
Rhône-Poulenc	RM 4.1, east side of LDW	shoreline	shallow	6.37	1.89
		shoreline	intermediate	6.64	0.02, 0.05
		shoreline	deep[g]	6.37 – 9.73	na
		275 ft from shoreline	shallow	6.38, 6.41	4.21, 6.63
		600 ft from shoreline	na	5.61 – 6.57[h]	na
South Park Landfill	approximately 2,000 ft southwest of the LDW	na[i]	deep	6.7[j]	na

Sources: Windward (2003c),Windward et al. (2005a), Ecology and Environment (2008b), Pacific Groundwater Group (2006), Pacific Groundwater Group (2007), Anchor (2006), SAIC (2007b), Environmental Partners (2008)

[a] River mile designations are approximate.

[b] Distances from the LDW shoreline are approximate.

[c] In the case of T-108 and Rhône-Poulenc, the aquifer zone was specified in the source report; in other cases, the aquifer zone was estimated based on well completion and screening depths and information on the aquifer characteristics of the Duwamish River Valley (Section 2.5.6.1).

[d] Distance reported is from the monitoring well location to the navigation channel of the LDW, rather than from the monitoring well to the RM 2.2 inlet.

[e] An anomalous DO value of 25.99 mg/L was reported but was excluded from this summary.

[f] Only one measurement was available for this property.

[g] The aquifer zone information available for these data indicated that the pH measurements were taken from wells located in the “lower” aquifer zone; this designation was interpreted as the deep aquifer zone.

[h] One pH reading for this well was not included in the range because it was noted in the source document to be lower than expected and the pH meter was recalibrated after this measurement was taken.

[i] The South Park Landfill is located approximately 2,000 ft southwest of the LDW.

[j] Represents a laboratory pH measurement rather than a measurement in the field.

DO – dissolved oxygen
LDW – Lower Duwamish Waterway
na – not available
RM – river mile
T-108 – Terminal 108

The mobility of organic compounds in groundwater is generally characterized by a retardation factor, which indicates the compounds' mobility relative to groundwater flow. Low-molecular-weight organic compounds (e.g., tetrachloroethene, 1,1,1-trichloroethane, BTEX) are the most mobile (retardation factors near 1), implying that chemicals with low soil-to-water partitioning coefficients travel close to the same rate as the groundwater. High-molecular-weight organic compounds, such as polychlorinated biphenyls (PCBs), have low mobility in groundwater and generally have retardation factors over 6,000 times slower than the movement of groundwater (Montgomery 1996). Conversely, in locations with high solvent concentrations, facilitated transport can decrease the retardation factor for hydrophobic chemicals.

The presence of dense non-aqueous phase liquid (DNAPL) can also affect groundwater quality. Upon release to the soil surface, free-phase DNAPL can move downward (via gravity) or laterally along the surface of soil layers. The DNAPL can become trapped in pores and fractures, leaving residual concentrations that may persist in the subsurface for many years. These residuals may slowly release chemicals into groundwater (EPA 1992a). No evidence of DNAPL was found in the LDW.

Light non-aqueous phase liquids (LNAPL) can also potentially affect groundwater quality, especially when soluble components of the LNAPL dissolve and create an aqueous phase plume (Newell et al. 1995). High groundwater flow rates and large contact areas between the LNAPL plume and the groundwater table may induce high dissolution rates of the plume. The LNAPL components may also undergo biodegradation within the subsurface and increase the formation of degradation products. While the soluble components of the LNAPL may dissolve and be transported in the aqueous phase in the subsurface, the less soluble components may be subject to volatilization and sorption. Parameters such as solubility, polarity, ionic charge, pH, and reduction-oxidation potential can all affect LNAPL sorption in the subsurface (Newell et al. 1995).

The mobility of dissolved metals in groundwater is influenced by their respective solubility, which is a function of pH, salinity, reduction-oxidation potential, and the presence of hydroxides and sulfide minerals in aquifer solids. In reducing environments, most metals are generally more mobile (USGS 1999), although their solubility can be decreased through precipitation reactions with groundwater anions (e.g., sulfate, hydroxides) if present in sufficiently high concentrations.

Metals in groundwater, such as copper and zinc, may have low mobility in the tidal mixing zone because of precipitation or sorption to oxides and clays, depending on the location of the reduction-oxidation boundary. Mixing of oxygenated estuarine water with groundwater within the aquifer results in higher concentrations of chlorides, sulfates, and carbonates in groundwater near the LDW. Reactions with these anions can lower the mobility of some metals (such as lead). Aerobic conditions in the tidally influenced zone also result in the formation of iron and manganese oxides (Boatman and Hotchkiss 1997), which provide an effective surface for sorption and co-precipitation with other metals in groundwater, thus limiting their discharge into estuarine systems (Moore et al. 1988).

The solubility and mobility of arsenic in groundwater is influenced by adsorption and desorption reactions, the presence of competing ions, and solid-phase precipitation and dissolution reactions (USGS 1996), which are a function of reduction-oxidation conditions and pH. An increase in dissolved arsenic concentrations might occur in areas natural organic matter has created a reducing environment, such as in shallow aquifers with high organic matter. Releases of organic chemicals such as TPH also cause reducing conditions as biodegradation processes consume available oxygen and cause the dissolution of arsenic (Moore et al. 1988). As groundwater becomes more aerobic (i.e., in the tidal mixing zone), arsenic may be oxidized to arsenate, which tends to sorb onto iron oxide coatings and clay minerals (USGS 1996). Increased oxygen concentrations in the tidal mixing zone can further attenuate arsenic.

2.6 SURFACE WATER HYDROLOGY

The Green River, which is the main freshwater source for the LDW, originates in the Cascade Mountains near Stampede Pass and flows by the Howard Hanson Dam at RM 65.0 and the Tacoma Headworks Dam at RM 61.0 (Culhane et al. 1995). Major tributaries to the Green River include Sunday Creek, Smay Creek, and the North Fork upstream of the Howard Hanson Dam and Newaukum Creek, Soos Creek, and Mill Creek downstream of the Howard Hanson Dam. In addition, the Black River discharges fresh water to the Duwamish River in Tukwila, several km upstream of the LDW. Flow from the Black River is normally minimal (approximately 2.6 m^3/s [92 cubic feet per second (cfs)]) but increases substantially during storm runoff events.

The lower 10 km (6.2 mi) of the river originally opened into a tidal marsh and a broad expanse of intertidal mudflats. In the mid-1800s, estimated annual average discharge from the Duwamish River ranged between 70 and 250 m^3/s (2,500 to 9,000 cfs) (Blomberg et al. 1988), although large floods, between 424.8 m^3/s and 849.6 m^3/s (15,000 and 30,000 cfs), used to down trees along the river's edge, deposit sediment throughout the river basin, and move river channels (Fuerstenberg et al. [1996], as cited in USACE 1997b). The Howard Hanson Dam was installed in 1961 in the upper part of the Green River primarily for flood control and low-flow augmentation to preserve fish life when river flows were naturally low (Sato 1997). The dam effectively decreased peak flows, which now rarely exceed 340 m^3/s (12,000 cfs) but increased moderate flows from 85 to 140 m^3/s (3,920 to 6,460 cfs) because of the periodic metered release of water stored behind the dam (King County 2000b).

Flow has decreased 78% from historical levels, mostly as a result of the diversion of the White River to the Puyallup River basin and the diversion of the Cedar River into southern Lake Washington in 1912. The former diversion was natural; the latter diversion was man-made in support of the construction of the Ballard Locks and the Lake Washington Ship Canal. These changes lowered Lake Washington by over 6 ft and caused increased drainage through the locks rather than through the Black River channel. Collectively, these irreversible changes have resulted in the present LDW hydrologic landscape.

The average downstream flow rate for the Duwamish River as measured at the Tukwila gaging station was 43.4 m^3/s (1,533 cfs) during 2003-2004 and ranged from 9.3 m^3/s (327 cfs) in August to 93.2 m^3/s (3,290 cfs) in June (Clemens 2007). Flow at the Auburn gaging station ranged from 4.3 to 329 m^3/s (152 to 11,600 cfs [the highest recorded flow rate since the construction of the Howard Hanson Dam]) between 1962 and 2004 (Clemens 2007). Average flow rates are based on mean daily values that were averages for the month (Clemens 2007). Between 2000 and 2006, the annual average flow rate measured at the Auburn gaging station was 33.7 m^3/s (1,190 cfs) (ranging between 24.1 m^3/s and 44 m^3/s [850.6 and 1,549 cfs]) (USGS 2007b). Approximately 80% of the water in the Duwamish River eventually flows through the West Waterway because of a sill at the south end of the East Waterway, which deters flow (Weston 1999a). Flow rates are greatest during the winter months because of seasonal precipitation and lowest throughout the late summer dry season.

Surface water runoff within the LDW area also contributes to flow within the LDW, from sources such as SDs, tributary creeks, CSOs, and non-point inputs; these sources are expected to be less than 1% of the total discharge, even during peak flow events. During the 1980s, SD discharges to the LDW were estimated at 1,868 million gallons per year (mgy) (0.2 m^3/s) (Tetra Tech 1988c). Annual CSO outflow to the LDW, derived from monitoring data collected between 2000 and 2005, averaged 75.7 mgy (0.008 m^3/s) (King County 2006a). The recent average SD inputs are small (approximately 0.02%) relative to the average flow of 43.4 m^3/s at the Tukwila gaging station during the 2003-2004 season.

Stream flow in the LDW is also influenced by water diversion, particularly by the City of Tacoma's Headworks Dam, which daily diverts at least 3.2 m^3/s (110 cfs) of fresh water for municipal use. Discharge of treated effluent from the King County South Wastewater Treatment Plant in Renton to the Duwamish River was eliminated in 1986, decreasing summer flows by as much as 25% (~1.6 m^3/s [56 cfs]) (Harper-Owes 1981).

EPA, the US Geological Survey (USGS), USACE, King County, Metro (now part of the King County Department of Natural Resources and Parks), the University of Washington, and other organizations have measured current velocities within the LDW as part of numerous environmental investigations (Harper-Owes 1983; King County 1999b; Prych et al. 1976; Santos and Stoner 1972; Stevens Thompson & Runyan 1972; Weston 1993b). The most extensive measurements within the LDW have been collected by King County. In 1996, King County deployed current velocity meters at two locations in the LDW (RM 1.1 and RM 3.5) for a 3-month period beginning in August, and recorded currents at 15-minute intervals along a vertical profile (King County 1999b). The two deployment stations were selected to represent the LDW study area with respect to channel width. The LDW is widest at RM 1.1 and is at its narrowest at RM 3.5. Measured current velocities within the LDW during this study rarely exceeded 40 cm/s (1.3 ft/s) (Lavelle et al. 1985).

In the winter of 2003-2004, King County conducted another study of current velocities, deploying two current velocity meters at RM 1.1 for two 4-week periods (King County 2005a). During both monitoring periods, one meter was placed near the center of the navigation channel, while the other was placed on a shallower channel side slope. Reported mean net current speeds for meters placed in the center of the channel ranged from 2.5 cm/s (at 25% channel depth) to 18 cm/s (at 10% channel depth). Mean net current speeds for meters placed at channel side slope locations ranged from 1.3 cm/s (at 25% depth) to 8.9 cm/s (at 10% depth). Currents were predominantly oriented along the channel, and velocities were generally slower along the side slopes. Current patterns corresponded to the semi-diurnal tide.

2.7 ESTUARINE FEATURES

Water circulation within the LDW, a well-stratified estuary, is driven by tidal actions and river flow; the relative influence of each is highly dependent on seasonal river discharge volumes. Fresh water moving downstream overlies the tidally influenced salt water entering the system. Typical of tidally influenced estuaries, the LDW has a relatively sharp interface between the freshwater outflow at the surface and saltwater inflow at depth.

Santos and Stoner (1972) characterized the circulation patterns within the tidally influenced water (or salt wedge) area of the LDW, which typically extends from Harbor Island to near the head of the navigation channel. When freshwater inflow from the Green/Duwamish River is greater than 28.3 m^3/sec (1,000 cfs), the saltwater wedge does not extend upstream beyond the E Marginal Way S Bridge (RM 6.3), regardless of the tide height (Stoner 1967). During high-tide stages and periods of low freshwater inflow from the Green/Duwamish River, the saltwater wedge has been documented as extending as far upstream as the Foster Bridge (RM 8.7) (Stoner 1967). At the river's mouth at the northern end of Harbor Island, a salinity of 25 parts per thousand (ppt) is typical for the entire water column; salinity decreases toward the upriver portion of the estuary. The thickness of the freshwater layer increases throughout the LDW as the river flow rate increases.

Dye studies indicate that downward vertical mixing over the length of the saltwater wedge is almost non-existent (Schock et al. 1998). Santos (1972) described how the upstream location or "toe" of the saltwater wedge, typically located between Slip 4 and the head of the navigation channel, is determined by both tidal elevation and freshwater inflow from the Green/Duwamish River. Fluctuations in tidal elevation also influence flow in the upper freshwater layer, which varies over the tidal cycle.

USGS measured the average net upstream transport of saltwater below the Spokane Street Bridge and reported it as approximately 5.4 m^3/s (190 cfs) (Clemens 2007). This average net upstream flow was about 12% of the average downstream flow measured at the Tukwila gaging station as noted in Section 2.6. During seasonal low-flow conditions, saltwater inputs from the West Waterway were more than one-third of the total discharge from the LDW (Harper-Owes 1983).

2.8 ECOLOGICAL HABITATS AND BIOLOGICAL COMMUNITIES

This section briefly summarizes the types of ecological habitats in the LDW, as well as the species that use these habitats, including benthic invertebrates, fish, wildlife, and plants. Additional information on the ecological habitats and biological communities of the LDW study area is provided in the ERA (Appendix A, Sections A.2.1 and A.2.2).

2.8.1 Ecological habitats and restoration activities

Ecological habitats of the LDW have been modified extensively since the late 1800s through hydraulic changes, channel dredging, the filling of surrounding floodplains, and the construction of overwater structures, levees, dikes, and other bank stabilization structures. The remnants of natural meanders along the LDW, several of which are now used as slips, and the area west of Kellogg Island are the only evidence of the river's original winding course.

The dominant natural habitat types in the LDW are intertidal mudflats, tidal marshes, and subtidal areas (Table 2-7). Most (around 98%) of the approximately 510 hectares (1,270 ac) of tidal marsh, 590 hectares (1,450 ac) of mudflats and shallows, and all of approximately 500 hectares (1,230 ac) of tidal wetland historically present in the LDW have been either filled or dredged (Blomberg et al. 1988). Remnant tidal marsh areas account for only 2 hectares (5 ac) of the LDW, while mudflats account for only 22 hectares (54 ac) (Leon 1980).

Table 2-7. Ecological habitat types represented in the LDW

HABITAT TYPE	DESCRIPTION	INFLUENTIAL PHYSICAL PARAMETERS	EXTENT AND CONDITION IN LDW
Intertidal marshes	Intertidal areas between -4 and +14 ft MLLW; exposed at low tides; marsh soils generally fine-textured and nutrient-rich, supporting grasses, sedges, rushes, and various other plants	salinity gradients, tidal variation, freshwater stream flow, wave action, water temperature, sediment characteristics, and oxygen content	limited extent; various marsh habitat classifications: emergent marsh (e.g., Herring's House, Hamm Creek, and Upper Turning Basin restoration areas), tidal marsh (5 ac in LDW); high and low marsh on Kellogg Island (Blomberg et al. 1988)
Intertidal mudflats	Intertidal areas roughly between -4 and +14 ft MLLW, remnant mudflats isolated from upland riparian vegetation, exposed at low tides, sometimes shaded by overwater structures	salinity gradients, wave action, water temperature, sediment characteristics, and oxygen content	flats and shallows (approximately 30 to 50 ac in LDW) (Blomberg et al. 1988); approximately 8.6 miles of exposed sand/mud substrate (Battelle et al. 2001); largest remnant on Kellogg Island
Intertidal riprap	Armored shoreline consisting of large rocks and rubble (riprap) or vertical wood or metal structures (sheet pile); can be anywhere from -4 to +14 ft MLLW.	salinity gradients, wave action, water temperature, and oxygen content	riprap-covered areas comprise approximately 17% of intertidal areas in the LDW (USFWS 2000b)

HABITAT TYPE	DESCRIPTION	INFLUENTIAL PHYSICAL PARAMETERS	EXTENT AND CONDITION IN LDW
Subtidal areas	Area deeper than -4 ft MLLW, never exposed by low tide; includes navigation channel and transition areas; sediment composition ranges from sand to mud	salinity, sediment composition, grain size, OC content, water depth, and temperature	throughout LDW, including navigation channel

LDW – Lower Duwamish Waterway
MLLW – mean lower low water
OC – organic carbon
USFWS – US Fish and Wildlife Service

The majority of the LDW shoreline consists of riprap, pier aprons, or sheet piling (Tanner 1991). Shoreline armoring is usually present at the top of the intertidal zone; areas of sloping mud, mudflats, and hard surfaces exist in the lower intertidal zone (Battelle et al. 2001). These hard surfaces support populations of encrusting organisms, such as barnacles, and burrowing organisms, such as shipworms (Leon 1980). Because of the shoreline armoring, the intertidal mudflats are partially isolated from inputs of sediment, nutrients, and organic matter (i.e., woody debris) from upland riparian vegetation zones; this isolation degrades the habitat quality of these mudflats (Battelle et al. 2001). In addition, overwater structures, which are common throughout the LDW, often shade shallow and intertidal habitats, alter microclimates, and inhibit growth of aquatic plant communities, further degrading the value of nearshore habitats for native fauna (Battelle et al. 2001).

Intertidal marshes are characterized by emergent aquatic plants and low shrubs, are tidally inundated, and occupy regional areas of the floodplain from approximately -4 to +14 ft MLLW (Blomberg et al. 1988; Bravo et al., undated draft). Marsh soils are generally fine-textured and nutrient-rich, supporting grasses, sedges, rushes, and various other plants. Intertidal marshes remain in the LDW to a limited extent, with some emergent marsh present at the Herring's House, Hamm Creek, and Upper Turning Basin restoration areas; approximately 5 ac of tidal marsh remain in the LDW, much of it on Kellogg Island (Blomberg et al. 1988). The shallow water and dense vegetation of intertidal marshes provide refuge as well as foraging and rearing habitat for benthic invertebrates and fish, including juvenile salmonids (Battelle et al. 2001). In addition, intertidal marshes provide important foraging and rearing habitat for many bird species, including great blue heron, killdeer, and marsh wrens. Taller vegetation, such as shrubs growing in intertidal marshes, can also provide nesting habitat for some bird species. In estuaries, intertidal marshes can also serve as wave buffers and provide flood attenuation (Battelle et al. 2001).

Intertidal mudflats are generally defined as the gently sloping areas from MLLW up to the edge of intertidal marsh vegetation (Blomberg et al. 1988). Intertidal mudflats in the LDW study area are located between -4 ft and +14 ft MLLW. These habitats can include mudflats consisting of unconsolidated silts and clays and sand flats consisting of unconsolidated sandy sediments (Simenstad et al. 1991). Approximately 30 to 50 ac of remnant intertidal mudflats and shallows (Blomberg et al. 1988), and approximately 8.6 miles of exposed sand/mud substrate exist in the LDW (Battelle et al. 2001). These areas occur on the northern portion of Kellogg Island and in occasional patches throughout the LDW, where they are exposed at low tides and sometimes shaded by overwater structures. Intertidal mudflats serve as sources of nutrients for primary producers, and provide food and habitat for benthic invertebrates, fish, shorebirds, and aquatic mammals. Intertidal mudflats also attenuate boat wakes for up-slope tidal marshes (Battelle et al. 2001).

Subtidal sediment composition ranges from sand to mud, depending on the sediment source and local current velocity (Windward 2003b). The sediments in the upstream portion of the LDW, near the head of the main channel at the Upper Turning Basin, are predominantly sand. The sediments in the subtidal habitat further downstream (e.g., near Kellogg Island) are characterized as brown or brown-gray sandy mud overlying darker, more clayey mud.

Kellogg Island, located to the south of Harbor Island, is the largest original habitat area remaining in the LDW and is currently designated as a wildlife refuge. Habitat associated with the island comprises high and low marshes, intertidal mudflats, and filled uplands. Kellogg Island is highly altered from its historical size, shape, and function as the result of dredge, fill, and diking activities that occurred from the late 1940s or early 1950s through the 1970s, which greatly altered its interior (Canning et al. 1979). The upland portion of Kellogg Island was created in 1974, when the Port of Seattle deposited 1,700 m^3 (2,200 cubic yards) of dredged material on the island (Sato 1997). Both introduced and native plant species rapidly colonized the 7-hectare (17-ac) island. Remnants of natural intertidal habitat occur on the northern portion of Kellogg Island and in occasional patches throughout the LDW.

Small intertidal areas of marsh and unvegetated marsh habitat in the LDW have become the focus of habitat restoration activities. The Hamm Creek restoration area, located on the western shoreline at RM 4.3, restored 1 ac of emergent salt marsh, 2 ac of freshwater wetlands, and nearly 2,000 ft of the Hamm Creek stream bed in order to provide suitable habitat for salmon and other fish and wildlife species (EPA 2006e). Habitat restoration projects are also located on Port property at Terminal 107 (T-107) (RM 0.4 west) and Terminal 105 (T-105) (RM 0.1 W) (Port of Seattle 2007b). At the Coastal America site in the Upper Turning Basin (RM 4.7 W), 5 ac of intertidal habitat was restored. At Herring's House Park, located on the west side of the LDW near Kellogg Island (RM 0.3 to RM 0.5 W), intertidal habitat has been restored at a former lumberyard.

Additional restoration activities are also taking place at the General Services Administration (GSA) marsh area (RM 0.8 E), the Diagonal Avenue S/Terminal 108 (T-108) restoration area (RM 0.6 E), the 8th Avenue S street end park/Gateway North restoration area (RM 2.7 E), the S Portland Street street end park (RM 2.8 W), the Muckleshoot Tribe restoration area at Kenco Marine near the Upper Turning Basin

(RM 4.6 W), the S 112th Street mitigation site (RM 5.7 E), and King County's Cecil B. Moses Park (North Winds Weir, RM 5.7 W). These restoration areas are presented on Map 2-8. Information on several additional restoration sites within the LDW is available through NOAA's Damage Assessment, Remediation, and Restoration Program (DARRP) (NOAA 2007b). The objectives of these projects include the removal of rock riprap and over-water wharf structures, restoration of natural tidal flows, and natural re-colonization by native wetland plants (Cordell et al. 1996).

2.8.2 Benthic invertebrates

Benthic invertebrate species are important components of the LDW ecosystem because they serve as a major food resource for commercially and recreationally important fish and wildlife and play a critical role in the overall nutrient cycling of the system. This section briefly describes studies that have investigated site use by benthic invertebrates. Additional details are presented in Appendix A, Section A.2.2.2.

2.8.2.1 Summary of LDW studies

Numerous studies have investigated site use of the LDW by benthic invertebrates. Over the course of several years, Cordell et al. (1996; 1997; 1999) conducted studies of the benthic community at several restoration sites throughout the LDW. Other studies of the benthic community were primarily conducted near Kellogg Island (NOAA and Ecology 2000; King County 1999a; Leon 1980; West et al. 2001; Williams 1990). Surveys of larger benthic invertebrates were conducted throughout the LDW in 2003 and 2004, including a survey of crabs and shrimp (Windward 2004a) and a survey of clams (Windward 2004c). In addition, sampling of the benthic community was conducted at both intertidal and subtidal locations throughout the LDW in 2004 as part of the RI (Windward 2005e). In 2006, Ecology conducted a sediment profile imaging (SPI) survey (Ecology 2007n) to identify relationships between SPI survey data and direct measurements of sediment quality. Metrics collected as part of the SPI survey were used to estimate the depth of the biologically active zone (BAZ) that was used in the RI. A detailed list of the taxa identified in these studies is included in Appendix A, Attachment A1.

2.8.2.2 Summary of site use

Benthic invertebrates in the LDW form two distinct communities. The infaunal community is typified by burrowing polychaetes and bivalves. King County (1999e) noted that at most sampling locations, the infaunal community was dominated by surface detrital/surface-deposit feeding organisms. The epibenthic community (invertebrates living on top of the sediment), consisted mainly of larger crustaceans and mussels, and was dominated by surface detrital and surface filter-feeding organisms.

Benthic invertebrate communities observed in the LDW consisted of 670 taxa, representing 178 families in 13 phyla. Typical of estuarine environments, the benthic invertebrate community was dominated by annelid worms (polychaetes and oligochaetes), mollusks, and crustaceans. Crustaceans were the most diverse of these three groups in the LDW, including more than 250 taxa. Mollusks included various bivalves and snails. The most abundant large epibenthic invertebrates included slender crabs, crangon shrimp, and coonstripe shrimp. Dungeness crabs were also common, although their distribution was generally limited to the portions of the LDW with higher salinity. Although the vast majority of benthic invertebrate species in the LDW are typical inhabitants of estuarine environments, a few organisms more typical of freshwater environments were found. For example, one chironomid larva was collected in intertidal habitat at RM 0.6, two chironomid larvae were collected in intertidal habitat at RM 1.4, and one chironomid larva was collected in the subtidal habitat at RM 1.6.

Key physical factors that may influence the distribution and abundance of benthic invertebrates in the LDW are salinity, water depth (intertidal versus subtidal), sediment grain size, and OC content. The LDW is a stratified estuary influenced by river flow and tidal effects. The daily salinity fluctuations favor species that are tolerant of such variability. A salinity range of approximately 5 to 8 ppt has been identified as the critical transition range that corresponds to a pronounced reduction of benthic invertebrate species richness (Levinton 1982). In general, benthic species richness diminishes steadily in an estuary until it reaches a minimum at the critical salinity level (Levinton 1982). A more diverse benthic invertebrate community generally exists in the downstream, more saline portion of the LDW (RM 0.0 to RM 2.0) (Windward 2005e). The benthic invertebrate community in the subtidal habitat is usually more diverse than the community in the intertidal habitat (Windward 2005e).

Common benthic communities in estuarine intertidal and subtidal channel habitats with mixed fines and muddy substrates are described in the classification system for marine and estuarine habitat types in Washington State (Dethier 1990). Benthic invertebrates typically present in intertidal habitats include the amphipods *Corophium salmonis*, *Paramoera columbiana*, and *Eogammerus* spp.; the polychaetes *Hobsonia florida* and *Manayunkia aestuarina;* the clam *Macoma baltica;* the shore crab *Hemigrapsus oregonensis;* tanaids; and mysids. In subtidal habitats, common invertebrates include a wide variety of polychaetes such as *Magelona* spp., *Capitella capitata*, P*araonella platybranchia*, *Eteone* spp., *Hobsonia florida*, and numerous spionids. Common subtidal crustaceans and bivalves are similar to the common species in intertidal habitats, with the inclusion of Dungeness crabs (*Cancer magister)* and shrimp (*Crangon* spp). The majority of these invertebrates or species within the same genus have been identified in the intertidal and subtidal habitats of the LDW.

Intertidal Community

Surveys of the intertidal benthic invertebrate communities (including infauna and smaller epibenthic species) were conducted at 12 locations throughout the LDW in 2004 (Windward 2005e). The total number of organisms ranged from 500 to 16,233 per

0.1 m^2 (Table 2-8). A total of 61 invertebrate taxa were identified in the sediment core and transect frame samples. The most abundant organisms were annelids and crustaceans. Numerous physical factors such as salinity, temperature fluctuation, desiccation, and wave action present physiological challenges to benthic invertebrates in the intertidal areas (Levinton 1982) and are thus contributing factors to the relatively low number of species in these areas. Some of these factors vary over the length of the LDW (e.g., salinity). Figure 2-10 presents the composition of major benthic invertebrate taxonomic groups at the intertidal locations as a function of river mile.

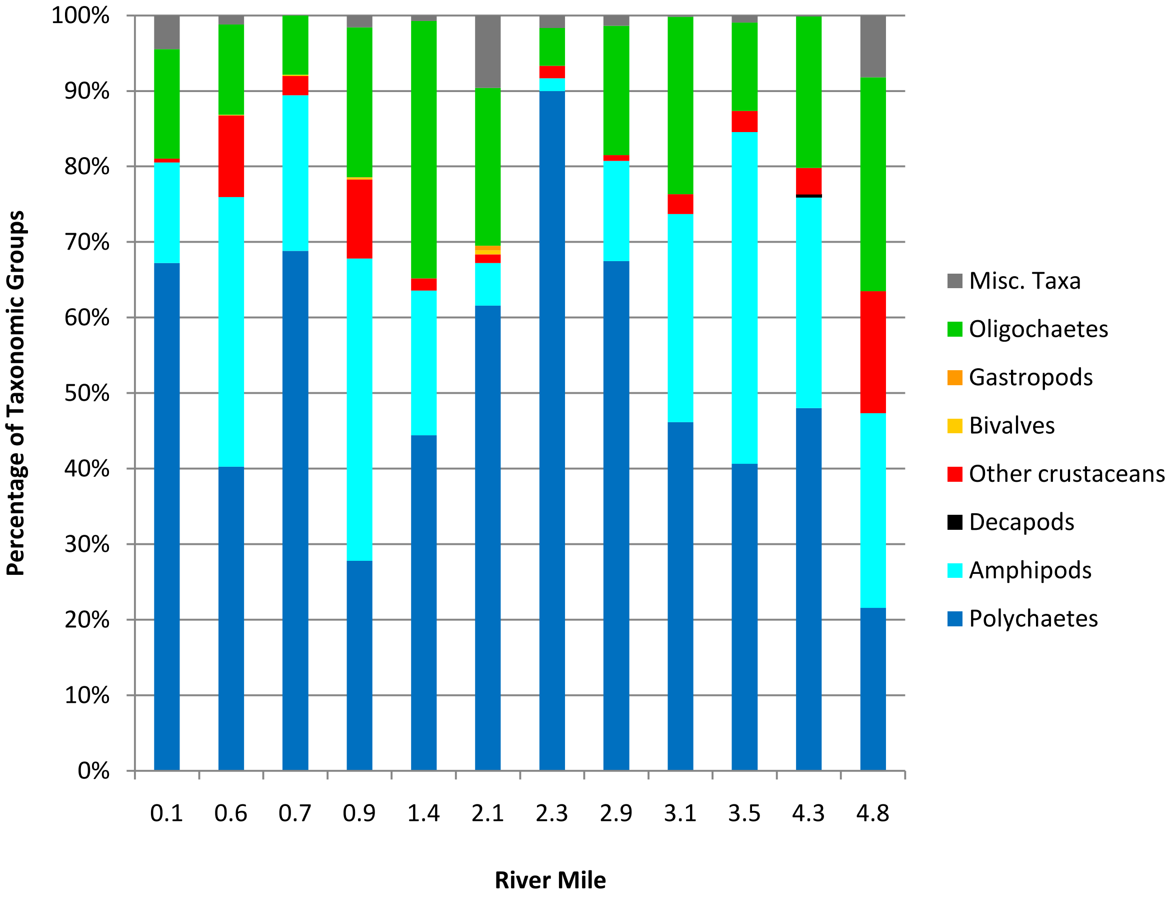

Figure 2-10. Composition of major benthic invertebrate taxonomic groups in intertidal areas of the LDW

Amphipods and polychaetes together represented more than 65% of the organisms identified at locations between RM 0.1 and RM 4.3. At the location in the Upper Turning Basin (RM 4.8), the benthic community was still estuarine; however, the composition had changed with amphipods and polychaetes together representing 48% of the organisms identified and three other groups (oligochaetes, other crustaceans, and miscellaneous taxa) representing the majority (51%) of the organisms identified. There are a few differences in the species composition in the intertidal community in the downstream versus the upstream end of the LDW. For example, species of

Cnidaria, including sea anemones, were only present in the intertidal areas from RM 0.0 to 1.0, and gastropods were present only at or downstream of RM 2.1. A Swartz's Dominance Index was calculated for each location. The Swartz's Dominance Index is defined as the minimum number of taxa that make up 75% of the total abundance (Swartz et al. 1985; as cited in PTI 1993). A higher number of taxa indicates a more diverse community; values of 5.0 or less are generally indicators of stressed communities (PTI 1993). In addition to the samples collected to assess the benthic community, separate samples were collected at each location and analyzed for biomass. Total biomass ranged from 0.1 g ww at location B5a-2 to 2.7 g ww at location B6a. Average biomass for intertidal locations was 1.4 g ww (Windward 2005e). Total weights at each location represented the biomass collected from an area of 0.012 m^2.

Cordell et al. (2001) conducted both epibenthic and infaunal surveys at seven restoration sites throughout the LDW from 1993 through 1999. The surveys demonstrated that the diversity and abundance of intertidal organisms varied both seasonally and among locations in the LDW. The greatest diversity of organisms (i.e., species richness) occurred in the downstream portion of the LDW; diversity was lower in the area of the Upper Turning Basin.

Two other benthic invertebrate surveys were conducted in the LDW by Williams (1990) and Leon (1980). Williams (1990) identified 80 invertebrate taxa inhabiting intertidal habitats at Kellogg Island. Nematodes, oligochaetes, small harpacticoid copepods, ostracods, and sabellid polychaetes were the dominant invertebrates. Leon (1980) found 43 different benthic taxa in sediment cores from the intertidal mudflats at Kellogg Island. Most taxa occurred infrequently; nine taxa accounted for 97% of all individuals. Small marine worms of the genus *Manayunkia*, oligochaetes, and harpacticoid copepods made up nearly 80% of all individuals (Leon 1980). In comparison, there were very few organisms at a mudflat site with anoxic sediments near the Duwamish Shipyard, (RM 1.35 W) and there was a greater degree of seasonal variability in the benthic community at a mudflat site at the Riverside Marina, then located near RM 0.2 W (Leon 1980).

In 2003, a clam survey was conducted at 11 intertidal locations between RM 0.0 and RM 4.0 (Windward 2004c). A random sampling design, based on Washington State Department of Fish and Wildlife (WDFW) guidance (Campbell 1996), was employed to survey each of the 11 locations for clam abundance. The mean density of clams at those 11 locations ranged from 0.18 to 0.94 per ft^2. The majority of clams collected were identified as *Macoma balthica* (60%), followed by *Mya arenaria* (20%) and *Macoma nasuta* (18%). *M. balthica* were the smallest clams collected on all beaches with a mean individual wet weight of 0.7 g (range of < 0.1 g to 3.4 g), and *M. arenaria* were the largest clams collected (mean individual wet weight of 15.2 g); however, this species also had the widest range in weight (0.3 g to 54 g) (Windward 2004c).

Table 2-8. Results of the intertidal benthic invertebrate community survey conducted for the RI

River Mile	Location ID[a]	Total Abundance[b]	Taxa Richness[c]	Swartz's Dominance Index	Taxa Richness by Major Taxonomic Group[d]				
					Annelida	Crustacea	Insecta	Mollusca	Misc. Taxa[e]
0.2	B1a	4,842	13	3	4	6	1	0	2
0.6	BCA-1[f]	11,667	22	6	7	9	2	1	3
0.8	B3a	11,958	25	4	9	13	0	2	1
0.9	B2a	16,233	30	8	8	12	2	3	5
1.4	B4a	8,858	23	4	6	13	2	0	2
2.1	B6a	1,475	19	5	8	5	0	2	4
2.3	B5a-2	500	10	3	6	2	0	0	2
2.9	BCA-3	9,050	16	2	8	6	0	0	2
3.1	B7a	8,600	20	4	6	11	1	0	2
3.5	B8a	14,100	21	6	7	11	0	0	3
4.5	B9a	5,875	24	5	8	13	1	0	2
4.8	B10a	6,600	20	6	7	10	0	0	3

[a] Sampling locations are shown on Map A.2-1 in Appendix A. BCA locations were sampled only for the benthic invertebrate community analysis.

[b] Total number of individual organisms in a standard 0.1-m^2 area determined by extrapolating the number in the composite samples, representing a total area of 0.012 m^2, to the number expected in the larger area (0.1 m^2) by multiplying by 8.33.

[c] Total number of taxa in a composite of five core samples, representing a total area of 0.012 m^2, at each location.

[d] Total number of taxa in each major taxonomic group in a composite of five core samples, representing a total area of 0.012 m^2, at each location.

[e] Miscellaneous taxa include Nemertea, Nematoda, Cnidaria, and Platyhelminthes.

ID – identification

RI – remedial investigation

Potential clam catch rates by people were assessed during the clam survey conducted in 2003 (Windward 2004c). Catch rate exercises were conducted at four beaches identified during the intertidal survey as having high clam abundance. The catch rate exercise used visible signs of clam presence such as holes or siphons to target where to dig, and also focused on collecting larger clams to simulate clamming behaviors that would most likely be conducted by harvesters. *M. arenaria* was the most common clam collected during the assessment of potential catch rates, comprising more than 98% of the total biomass.[14] Catch rates (i.e., total number of clams caught per hour) for all beaches combined were 652 for *M. arenaria* and 189 for all other species.

Subtidal Community

Surveys of the subtidal benthic communities (including infauna and smaller epibenthic species) were conducted at 14 locations throughout the LDW in 2004 (Windward 2005e). The total number of organisms ranged from 72 to 2,300 per 0.1 m^2 (Table 2-9). A total of 246 invertebrate taxa were identified in the van Veen grab samples. In general, annelids, crustaceans, and mollusks were the most abundant organisms at the subtidal locations. Benthic species richness in the subtidal habitats generally diminished with distance from the mouth of the river as the habitat changed from predominantly marine to estuarine. The largest decrease in species richness occurred between RM 0 and RM 3.0. Figure 2-11 presents the composition of major benthic invertebrate taxonomic groups at the subtidal locations as a function of river mile.

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[14] The results of the catch rate exercise indicated that although *M. balthica* was the most abundant clam found in the quantitative survey, *M. arenaria* was the clam most likely to be harvested because of their larger size and presence of shows.

Table 2-9. Results of the subtidal benthic invertebrate community survey conducted for the RI

RIVER MILE	LOCATION ID[a]	TOTAL ABUNDANCE[b]	TAXA RICHNESS[c]	SWARTZ'S DOMINANCE INDEX	TAXA RICHNESS BY MAJOR TAXONOMIC GROUP[d]				
					ANNELIDA	CRUSTACEA	ECHINODERMATA	MOLLUSCA	MISC. TAXA[e]
0.1	B1b	326	107	15	43	22	4	28	10
0.6	BCA-4	937	107	13	62	12	2	25	6
0.9	B2b	537	92	7	43	17	1	23	8
1.0	B3b	559	93	12	54	14	2	22	1
1.4	B4b	521	78	8	39	11	3	19	6
1.5	B5b	643	60	8	34	11	0	11	4
1.5	BCA-5	72	50	15	27	7 (1)[f]	0	12	3
1.7	BCA-2	328	54	6	29	12	1	11	1
2.2	B6b	1,137	83	3	41	17	0	21	4
2.7	B7b	497	75	5	38	16	1	18	2
3.9	B9b	935	36	6	17	12	0	6	1
4.2	B8b	2,300	27	4	14	12	0	1	0
4.3	B10b	1,541	16	2	9	7	0	0	0
4.6	BCA-6	1,689	14	3	7	7	0	0	0

[a] Sampling locations are shown on Map A.2-1 in Appendix A. BCA locations were sampled only for the benthic invertebrate community analysis.

[b] Total number of individual organisms retained on a 1-mm sieve in a standard 0.1-m^2 area determined by extrapolating what was enumerated in three composite van Veen grab samples, representing a total area of 0.3 m^2, to the smaller area (0.1 m^2) by dividing by 3.

[c] Total number of taxa in a composite of three van Veen grab samples, representing a total area of 0.3 m^2, at each location.

[d] Total number of taxa in each major taxonomic group in a composite of three van Veen grab samples, representing a total area of 0.3 m^2, at each location.

[e] Miscellaneous taxa include Nemertea, Nematoda, Cnidaria, and Platyhelminthes.

[f] One insect specimen was also collected at this location.

ID – identification

RI – remedial investigation

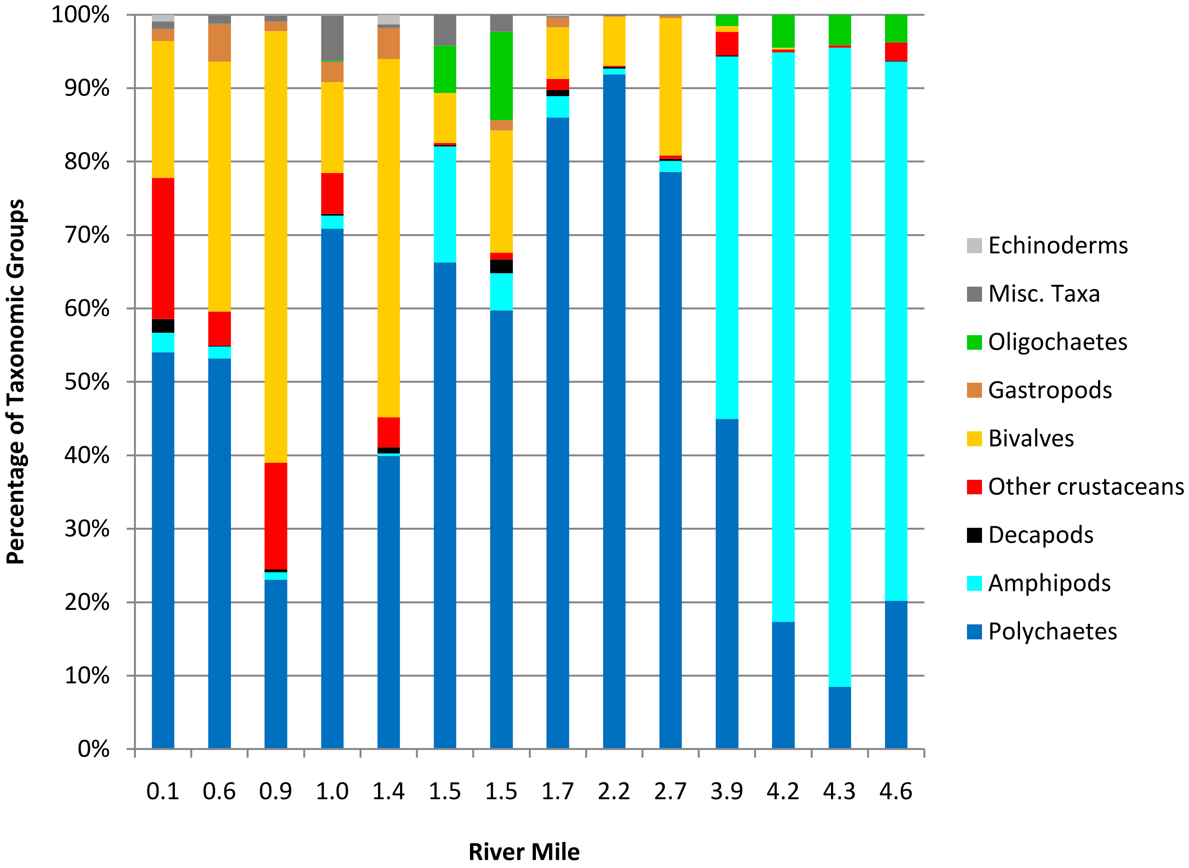

Figure 2-11. Composition of major benthic invertebrate taxonomic groups in subtidal areas of the LDW

Bivalves and polychaetes together represented more than 70% of the organisms identified at locations between RM 0.1 and RM 2.7. A clear shift in species composition occurred between RM 2.7 and RM 3.9. Bivalves were absent from the benthic community at the upstream locations (RM 3.9 to RM 4.6), and polychaete abundances at the three upper-most locations (RM 4.2 to RM 4.6) were reduced to 20% or less of the organisms identified. Amphipods were a dominant taxonomic group, accounting for 49 to 87% of the organisms at each upstream location (RM 3.9 through RM 4.6); whereas at downstream locations, amphipods accounted for a maximum of 16% of the organisms identified. Some of the main differences at the species level at upstream locations (relative to downstream locations) were: 1) the absence of butterclams, littlenecks, and cockles (these bivalves were relatively small in size and primarily found in the subtidal areas); 2) the absence of some marine polychaete genera, such as *Nephtys* and *Phyllodoce;* and 3) an increase in the abundance of some species, such as the amphipods *Americorophium salmonis* and *A. spinicorne* and the polychaete *Hobsonia florida*. These species are common in estuarine habitats with large salinity fluctuations. Additional differences in species composition at upstream locations (relative to downstream locations) were: 1) a reduction in the number of decapod species to one

(*Crangon franciscorum*, a species known for its tolerance of low salinity (Jensen 1995)); 2) the disappearance of gastropods, echinoderms, and miscellaneous taxa at locations at or above RM 3.9; and 3) a more consistent presence of oligochaetes at and above RM 3.9. The Swartz's Dominance Index calculated for each location supported this trend of diminishing numbers of taxa in upstream locations. Similar to the intertidal locations, samples were also collected and analyzed for biomass. Total biomass ranged from < 0.1 g ww at locations B7b, BCA-4, and BCA-5 to 1.7 g ww at location B4b. Average biomass for subtidal locations was approximately 0.34 g ww (Windward 2005e). Total weights at each location represented the biomass collected from an area of 0.0072 m^2.

Ecology (2000) evaluated the benthic community at three subtidal locations in the LDW as part of a sediment quality reconnaissance study for central Puget Sound. The benthic communities at the three locations were dominated by annelids. Mollusks were also common, whereas crustaceans and echinoderms were present in low abundances.

King County evaluated abundance of benthic infauna and epibenthos as a component of their assessment of CSO discharges to the LDW and Elliott Bay (King County 1999e). Subtidal sampling sites included transects located at Kellogg Island as well as downgradient from the Diagonal Avenue S CSO/SD and Duwamish CSO. Polychaetes were abundant in all samples and were the dominant organisms at all locations except at two locations downstream of the Diagonal Avenue S CSO/SD and Duwamish CSO, where oligochaetes and mollusks were dominant. A Kellogg Island location also had relatively abundant mollusks. Arthropods tended to be more abundant in deeper waters (King County 1999e).

Williams (1990) sampled epibenthic biota to a depth of –8 ft MLLW near Kellogg Island and reported that nematodes, oligochaetes, small harpacticoids, and cumaceans dominated the subtidal epibenthos. As with the intertidal benthos, stations with finer sediments generally had a greater abundance of epibenthic biota (Williams 1990).

Leon (1980) used van Veen grab samplers to characterize the subtidal epibenthic and infaunal sediment biota at five locations downstream of RM 1.5. The study identified more than 60 different taxa, greater than the number found in the intertidal habitat from the same survey. Most subtidal species were deposit-feeding polychaete worms characteristic of deeper, turbid waters of the LDW. Small deposit-feeding clams (*Macoma* sp., *Axinopsida* sp., and *Psephidia* sp.) and the amphipod *Anisogammarus* sp., which feeds on diatoms and green algae, were also present.

Larger Epibenthic Invertebrates

Larger epibenthic invertebrates identified in the LDW include crabs, shrimp, sea stars, anemones, and mussels. Numerous larger epibenthic invertebrate species were caught during two fish and crab surveys conducted throughout the LDW for the RI (Windward 2005b, 2006c). The most abundant epibenthic invertebrates were slender crabs, crangon shrimp, and coonstripe shrimp. Dungeness crabs were also caught in both surveys, although their distribution was generally limited to the portions of the LDW with greater salinity.

A pilot survey and three quarterly surveys were performed in 2003 and 2004 to estimate the abundance of crabs and shrimp within the LDW (Windward 2004a). Three crab species and one shrimp species were caught during the surveys. Slender crabs and Dungeness crabs were the most abundant species, with red rock crabs and dock shrimp being less abundant (Windward 2004a). The majority of these crabs were caught in the downstream, more saline part of the LDW. Red rock crabs and dock shrimp were also caught, primarily in the downstream portion of the LDW, with a few adults caught between RM 1.6 and RM 2.2. During a reconnaissance study conducted by Environmental Solutions Group, Inc. (ESG) (1999), Dungeness crabs and red rock crabs were found at multiple locations near Kellogg Island; no adults were collected upstream of this point, whereas juveniles were found up to the 1st Avenue S Bridge.

During the Puget Sound Ambient Monitoring Program (PSAMP) (1989 to 1999), invertebrates were collected throughout Puget Sound, including locations in the LDW (West et al. 2001). Epibenthic invertebrate species, similar to those caught in the surveys conducted for the RI, were collected with otter trawls. The most common invertebrates were slender crabs and crangon shrimp. Other species caught during the PSAMP survey, but not in the RI surveys, included porcelain crabs (family Porcellanidae), chitons (class Polyplacophora), and several sea stars.

In September of 2006, King County collected fish for tissue analyses using a high-rise otter trawl in the downstream portion of the LDW (Anchor and King County 2007). Non-target invertebrate species were recorded and released. Coonstripe shrimp was the most common invertebrate species, followed by graceful crab and crangon shrimp. Other crab species included kelp crab, decorator crab, red rock crab, black-clawed crab and pygmy rock crab. Sea star species include short-spined sea star, sunflower sea star, and false ochre star. Plumose anemone was also found during this study.

Biologically Active Zone

The BAZ refers to the surface sediment layer where sediments are mixed by the feeding and burrowing behaviors of benthic invertebrates. This layer is often the focus of ERAs for benthic invertebrates and fish and is the point of compliance for surface sediments under the Washington State Sediment Management Standards (SMS). According to Ecology's sampling and analysis guidance (Ecology 2008c), "Past studies in Puget Sound have demonstrated that the majority of benthic macroinvertebrates are generally found within the uppermost 10 cm of the sediments. While some species may be found at deeper depths below the sediment surface, 10 cm is generally assumed to represent a reasonable estimate of the biologically active zone." This section provides a discussion of site-specific data collected during an Ecology-led investigation of SPI techniques for assessment of benthic communities in the LDW and elsewhere (Ecology 2007n) and a King County-led study of the Duwamish/Diagonal dredging residuals area prior to the placement of a thin sand layer (Anchor 2007) to determine the appropriateness of the assumption that 10 cm represent the BAZ in the LDW RI/FS.

Background

Benthic organisms usually inhabit the oxygenated layer[15] of the sediment near the sediment-water interface. The depth that organisms occur within the sediment column is related to the habitat characteristics (grain size, hydrologic/sediment transport regime, amount and quality of food) and degree of perturbation that a given location experiences (e.g., periodic erosion from high currents or propeller scour; predation by larger organisms such as fish or crabs; exposure to chemical contamination in water, sediment, or food). Benthic communities in stable environments with adequate food and sediment quality typically include larger, less abundant, longer-lived organisms (e.g., deposit-feeding polychaetes) that burrow and feed up to 20 cm below the sediment surface (Germano & Associates 2007).[16] This burrowing depth is often below the oxygenated layer at the sediment surface; however, these mature (Stage 3) communities employ physical or biological adaptations that give them access to the oxygenated water at the sediment surface and retard the influx of anoxic porewater into their burrows or tubes. In stable, depositional areas of the LDW where Stage 3 communities would be expected to be common, it is likely that organisms occasionally burrow and/or feed in the 10–to-20-cm stratum of the sediment.

Benthic communities that undergo periodic or chronic disturbances (physical or chemical) are more likely to be characterized by very small, abundant organisms (typically opportunistic polychaetes and bivalves) that take advantage of the short-term availability of a habitat or a resource. These early colonizers (Stage 1 assemblages) live within a few millimeters of the sediment-water interface, reflecting both their small size and the depth to which oxidized sediment occurs, primarily through the molecular diffusion of oxygen from the overlying water column. Disturbed communities that have undergone some type of recovery will have infaunal members (e.g., tubicolous polychaetes, amphipods, small bivalves) that may burrow to shallow (within a few centimeters) depths in the sediment column. These transitional communities (Stage 2 assemblages) are the first to rework subsurface sediment and extend the oxygenated sediment layer to several centimeters or more. Regardless of the successional stage, the oxygenated sediment layer seldom extends deeper than about 10 cm.

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[15] Depth of the oxygenated layer is primarily a function of advection, diffusion, biological activity, and physical energy. Diffusion is the least effective, on the order of a few mm. Organisms, through feeding and burrowing, open up spaces and can actively ventilate oxygenated water down into the sediment. Physical energy (e.g., intertidal swash zones) can drive oxygenated water into the sediment. The depth varies depending on the type of organisms present, among other factors.

[16] There are individual organisms (e.g., various bivalves, ghost shrimp, nemerteans, polychaetes) in specific environments (e.g., quiescent environments) that can occur at deeper sediment depths.

SPI can be used to estimate the depths of the oxygenated layer and document depths of macroinvertebrate tubes or feeding voids. The SPI camera is deployed multiple times at a location and collects three replicate images of the sediment column approximately 15 cm wide and up to 20 cm deep. Each image is evaluated using image analysis software, with visual confirmation of the results. In many SPI surveys, plan view photos are also collected and are used to characterize the physical regime and other biological characteristics associated with the profile images.

Results

SPI data were collected at 87 stations in the LDW as part of an Ecology-led study in July 2006. The purpose of this study was to assess the relationships between SPI metrics and direct measures of sediment quality (sediment chemistry, sediment toxicity, and benthic community assemblages, as defined by the SMS). The study targeted communities in fine-grained sediments in areas with a range of chemical contamination between RM 0 (located at the southern end of Harbor Island) and RM 3.4. The SPI sampling was limited to highly saline subtidal areas with a minimum depth of 8 ft below MLLW. As part of the Ecology study, 23 different attributes were quantified for each image, including the following key attributes related to the assessment of the BAZ and distribution of community types that strongly influence the BAZ depth: the apparent redox potential discontinuity (aRPD) depth that indicates the *approximate* depth of the oxygenated layer; apparent grain size (based on phi classification); number and depth of observable subsurface voids; boundary roughness[17] and type (biological or physical); small-scale bed dynamics (depositional, erosional, or static); and successional stage of the benthic community (Germano & Associates 2007).

King County conducted a similar investigation of baseline conditions prior to the placement of a thin layer of sand in areas with residual contamination following dredging in the vicinity of the Duwamish/Diagonal CSO/SD in 2005 (Anchor 2007). SPI images were collected from 28 stations and images were minimally processed; only the average aRPD and void depth were reported. Results of these two studies are discussed in the following sections, by SPI metric.

Oxygenated Layer Depth — The apparent depth of the oxygenated layer (i.e., aRPD) is a good indication of where Stage 1 and the majority of Stage 2 organisms may be found in the sediment column. The aRPD was reported as the minimum and maximum depths measured in each replicate image, along with an average depth for each image. The depth of this feature across all images from the Ecology study ranged from 0.2 to 9.6 cm below mudline and averaged 2.6 cm (the 95% upper confidence

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[17] Boundary roughness is the vertical distance between the highest and lowest points of the sediment-water interface over the width of a single image. This small-scale surface relief ranged from 0.2 to 9.1 cm in the LDW and was related to either physical sediment structures (e.g., ripples) or biogenic features (burrow openings, fecal mounds, foraging depressions).

limit on the mean [UCL] was 2.8 cm). The average aRPD (i.e., the average of three replicate image averages) at all Ecology sampling locations is presented on Map 2-9. The aRPD in the vicinity of the Duwamish/Diagonal CSO/SD was similar to that reported in the Ecology study, averaging 2.4 cm (with a 95% UCL of 2.7 cm).

Voids — Voids are subsurface oxygenated spaces in the sediment column created by organisms burrowing or ingesting sediment. These activities mix the surface sediments with subsurface sediment and are the major contributor to the process known as bioturbation. Void depth can be used as an indication of the maximum depth of the sediment column in which benthic invertebrates exist and may be exposed to sediment chemicals.[18]

Evidence of voids was found in half of the replicate images (119 out of 238 profile images) evaluated in the Ecology study. Where present, only a few (1 to 3) voids were typically noted per image, although in a dozen cases the number ranged from 4 (n = 7) to a maximum of 8 (n = 1) voids.[19] Void depths were reported as an interval (minimum/maximum) in the sediment within which voids occurred (if only one void was present, then the interval represented the top and bottom of that void). Depths of voids ranged from 0.5 to 19.5 cm below the sediment surface among all of the replicate images with voids, indicating that some benthic organisms in the LDW actively burrow or feed at depths of 10 to 20 cm in the sediment column. The average void depth per replicate was 10.3 cm (the 95% UCL was 11 cm) in fine-grained sediments, with about half of the void bottoms occurring at depths between 10 and 20 cm below the sediment surface. The minimum and maximum void depths at each station (three replicates combined) are provided on Map 2-10. These findings were similar to those reported in King County's study, which reported the average depth of bioturbation as ranging from 1.8 cm to 15.3 cm (the overall average was 8.4 cm; the 95% UCL was 9.4 cm).

Where voids were present, the maximum void depth was greater than the maximum aRPD in almost all cases. Voids below the oxygenated sediment layer were distributed throughout the study area (i.e., RM 0 to RM 3.4); no particular spatial pattern was found other than an association with fine-grained sediment.

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[18] Tube-building organisms and those that construct permanent (or semi-permanent) burrows limit their exposure by sealing tubes or lining burrows with mucous.

[19] The proportion of the cross-sectional area that is void space can be estimated for each image. However, without other measures made over time, the proportion is not a good estimator of absolute exposure or bioturbation; but it may be useful for indicating a relative rate of biological activity among locations. The cross-sectional areas of the voids were estimated to be between 0.1 and 20 cm^2 (based on a sampling of 25% of the replicate images with a single void). As a proportion of the surface area of the sediment in the profile image, the areal extent of sediment disturbed by void-producing activities ranged from < 1 to about 10% of a given image. Areal extent was calculated as the product of the number of voids and the estimated average void area divided by the area of the profile image reported by Ecology (2007n).

Boundary Roughness (Physical Feature) — Roughness was measured as the vertical distance (or amplitude) between the highest and lowest mudline elevations within a profile image. The maximum roughness was about 9 cm and averaged 1.1 cm. A higher measure is indicative of physical disturbance, such as current-induced ripples on the sediment bed; a lower measure (generally less than 3 cm) indicates mostly biological disturbance of the sediment bed. This information can be used to help characterize the physical regime to which the benthic communities are exposed and the likely community assemblages that may occur in response to the physical environment.

Small-Scale Sediment Bed Dynamics — Information about small-scale sediment bed dynamics can be used to predict, in part, the benthic community structure that may be found in different areas of the LDW. Features in both the plan and profile images that were indicative of various physical and biological processes were noted in Ecology's report. These features included, but were not limited to, sediment sorting and fabric and the presence of bedforms, faunal structures, and debris. From combinations of such features, Ecology inferred the extent to which a station is likely erosional or depositional at small spatial scales (profile images were about 15 cm wide; plan view images were about 50 cm wide). Over 70% of the stations examined were classified as depositional; 18% were classified as erosional; 8% were static (i.e., no net loss or accumulation of sediment); and 2% of the stations were not classified because their classification could not be determined based on the SPI metrics. One caveat is that no temporal scales were evaluated so these classifications may not be indicative of long-term conditions.

Community Successional Stages — An examination of successional stage classifications provided in Ecology's report showed that 60% of the replicate images evaluated were indicative of mature (Stage 3) communities, 28% of the images were indicative of a transitional community (Stage 2), and 4% of the images were indicative of early colonizing communities (Stage 1); the community stage could not be determined in 8% of the images.[20] At the stations where Stage 3 communities were present, there was often a mixture of Stage 1 or 2 communities as well; this was interpreted as evidence of recruitment/growth in a mature community or very small-scale perturbations. Community stage also varied among replicates at 40% of the sampling locations (e.g., one image showed a Stage 2 community, and two images showed Stage 3 communities); this variability is likely indicative of the small spatial-scale heterogeneity that benthic communities tend to exhibit in response to the biological and physical features of their environment. Map 2-11 displays the typical (based on two or more replicates) successional stage at each sampling station.

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[20] Indeterminate classification was usually the result of insufficient penetration of the camera (substrate was compacted coarse-grained material), or when algae or wood that was entrained by the camera into the sediment obscured the camera's view.

The distribution of successional stages appeared to be strongly linked to sediment grain size and small-scale sediment bed dynamics (Table 2-10). In quiescent or depositional areas, sediments tend to be fine-grained and can support deeper burrowing, more mature communities. In general, Stage 3 communities (whose members are responsible for the deeper voids) are expected to dominate depositional areas. Conversely, early colonizers (Stage 1) or transitional communities (Stage 2) are expected to populate areas where sediments (typically sands) are being actively transported because of the instability of the habitat. As shown in Table 2-10, over 90% of the Stage 3 communities were found in fine-grained depositional areas (as expected), and about 40% of the early colonizing or transitional communities were found in areas noted in Ecology's report as having evidence of erosion or sediment transport at this small scale.[21] Based on the location of the transitional communities that were present in fine-grained areas (i.e., where mature communities would be expected), ship traffic (along pier faces, areas of tug activity, etc.) or other factors (e.g., located within or on the slope of the navigation channel; presence of sediment chemical contamination) may be contributing to small-scale or periodic perturbations that are affecting community succession. Table 2-10 presents the distribution of benthic community successional stages by physical regime, as defined by the Ecology SPI study (2007n).

Conclusions

The default assumption of 10 cm can be reasonably applied as the BAZ in the LDW based on the following factors:

- **Representativeness of the entire benthic invertebrate community –** Overall, the SPI data indicate that although individual benthic invertebrates may be utilizing depths between 10 and 20 cm below the sediment surface, the benthic invertebrate community primarily utilizes shallower sediment based on the limited number of voids below 10 cm, the limited areal extent of deeper sediment disturbed by feeding or burrowing activities, and the prevalence of Stage 1 and 2 organisms that co-occur with the Stage 3 communities.
- **Relationship between voids and depth in sediment –** The number of voids drops off dramatically with distance from the sediment surface. Specifically, where voids were observed, 49% were contained in the 0–to-10-cm interval, 38% of the voids were contained in the 10–to-15-cm interval, and only 13% of the voids extended to depths greater than 15 cm.
- **Central tendency of void depth –** The mean of the maximum void depths observed in individual profile images was approximately 10 cm (with a

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[21] Erosion/transport classification is based on evidence of layering, sorting, and/or bedding in SPI images.

95%UCL of 11 cm), which was corroborated by the King County study (Anchor 2007).

- **Location –** Other than an association with fine-grained sediment, there were no apparent clusters of areas with deeper voids; voids seem to be distributed throughout the study area and not related to presence or absence of contamination.

Table 2-10. Distribution of benthic invertebrate community successional stages by physical regime in sediment profile images from the LDW

COMMUNITY STAGE	REPLICATE DISTRIBUTION BY SPI GRAIN SIZE AND PHYSICAL REGIME[a]					EXPOSURE	
	FINE-GRAINED DEPOSITIONAL	COARSE-GRAINED EROSIONAL OR TRANSPORT	MIXED GRAIN SIZE/STATIC	UNKNOWN[b]	TOTAL	MAXIMUM VOID DEPTH (cm)	MEAN DEPTH OF BOTTOM OF VOID (cm)
Early colonizers (Stage 1)[c]	0	6	0	0	6	0	0
Transitional (Stage 2)[c]	33	28	12	1	74	13.3[d]	10.6[d]
Mature (Stage 3)	143	3	9	3	158	19.5	10.3[e]
Indeterminate	11	11	0	0	22	na	na
Total	**187**	**48**	**21**	**4**	**260**	na	na

[a] SPI classification of physical regime is conceptually similar to the sediment dynamics discussed in Section 3; however, the SPI analysis represents a much smaller scale of assessment and does not reflect long-term trends. For example, an area identified as being potentially subject to erosion in the discussion of sediment dynamics in Section 3 might experience erosion infrequently (e.g., during episodic periods of high flows); in between such periods of erosion, the area may be subject to net sediment deposition. Hence, there may not be a direct correspondence between the physical regimes identified in the one-time SPI study and the physical regimes identified in Section 3 based on long-term conditions.

[b] Unknown physical regime was a sampling artifact (e.g., camera and/or frame disturbed the sediment surface or entrained debris into the sediment, precluding assessment of the sediment characteristics related to physical regime).

[c] This classification refers to samples where only this stage was present. Stage 1 and 2 organisms co-occurred with Stage 3 organisms throughout the study area and were considered evidence of recruitment or very small scale (< 1 cm) disturbance at that location.

[d] Voids were observed in only two replicate samples (one void per replicate); voids were absent in all other Stage 2 locations.

[e] Mean of the reported bottom void depth, where Stage 3 communities were present.

LDW – Lower Duwamish Waterway

na – not applicable

SPI – sediment profile imaging

SPI images in the LDW indicate that benthic invertebrates in Stage 3 communities actively burrow and feed between 0 and 20 cm below the sediment surface. Benthic invertebrates may occasionally exist below 10 cm in Stage 2 transitional communities.[22] Successional Stage 3 communities were the most abundant community type in the Ecology study area (RM 0 to RM 3.4) and typically co-occurred with Stage 1 and Stage 2 organisms. Communities in these earlier stages generally utilize the top 10 cm, as evidenced by the depth of the aRPD and the depths of voids where only Stage 1 or Stage 2 communities are present.

2.8.3 Fish

A diverse fish community inhabits the LDW. This section briefly summarizes what is known about the LDW fish community. Further details regarding life history characteristics and dietary preferences are presented in Appendix A, Section A.2.2.3.

2.8.3.1 Summary of LDW fish studies

Data on site use by fish are available from 14 studies that have been conducted or are ongoing in the LDW (see Appendix A, Table A.2-8, for a complete list of these studies). In combination, these studies have used various capture techniques to assess the diversity and abundance of fish in the LDW.

2.8.3.2 Summary of site use by fish

The LDW is inhabited by numerous anadromous and resident fish species (see Appendix A, Attachment A2, for a complete list of identified species). Fifty-three resident and non-resident fish species were captured in the LDW during sampling events conducted for the RI (Windward 2004d, 2005b, 2006c). During previous sampling events, Warner and Fritz (1995) recorded 33 resident and seasonal species of fish in the LDW, Miller et al. (1977a) observed 29 species, and Matsuda et al. (1968) recorded 28 species.

Simenstad (1983) identified 43 species of demersal fish commonly found in estuaries throughout the Pacific Northwest, Puget Sound, and the Pacific Ocean. Of the 43 species listed, 24 have been reported in the LDW. Species listed in Simenstad (1983) but not identified in the LDW include Pacific lamprey, green sturgeon, white sturgeon, common carp, leopard dace, white crappie, black crappie, yellow perch, redtail surfperch, walleye surfperch, spotfin surfperch, silver surfperch, kelp greenling, rock greenling, lingcod, cabezon, pricklebreast poacher, showy snailfish, and ringtail snailfish. Differences in the fish community structure between the LDW and other Pacific Northwest estuaries are likely a function of differences in salinity profiles, availability of specific habitat types, and sampling methods and seasons.

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[22] Two replicates categorized as transitional (i.e., with a Stage 2 community) each had a single void; voids were absent in all other Stage 1 and Stage 2 locations.

Based on the LDW fish studies reviewed in the ERA, shiner surfperch, snake prickleback, Pacific sandlance, Pacific staghorn sculpin, longfin smelt, English sole, juvenile Pacific tomcod, pile perch, rock sole, surf smelt, three-spine stickleback, Pacific herring, and starry flounder were identified as abundant at the time of the sampling events, as were chinook, chum, and coho salmon (see Appendix A, Attachment A2). Fish abundance in the LDW reaches its maximum in late summer to early fall and is generally lowest in winter (Dexter et al. 1981; Miller et al. 1977a). Based on 2004, 2005, and 2007 trawl data (Windward 2004d, 2005b, 2006c), differences between the fish community structure in the downstream (RM 0.0 to RM 2.4) and upstream (RM 2.9 to RM 5.0) segments of the LDW were apparent. Species diversity in the downstream segment was greater than in the upstream segment, with some species being rare or absent from the most upstream segment (RM 4.2 to RM 5.0) sampled for the RI. For example, English sole and rock sole were more abundant downstream, and starry flounder and shiner surfperch were more abundant upstream (Table 2-11). The salinity of the upstream portion of the river is lower than the salinity in the downstream portion, which may contribute to some of these differences in abundance.

Table 2-11. Catch per unit effort for fish species in the four fish sampling areas in the LDW

	AVERAGE NUMBER OF FISH PER TRAWL			
SPECIES	AREA T1 (RM 0.2 to RM 1.0)	AREA T2 (RM 1.6 to RM 2.4)	AREA T3 (RM 2.9 to RM 3.7)	AREA T4 (RM 4.2 to RM 5.0)
English sole	4.7	17.6	4.9	2.3
Longfin smelt	0.7	< 0.5	2	2.9
Pacific herring	< 0.5	1	< 0.5	< 0.5
Pacific staghorn sculpin	2.8	8.2	5.2	11
Pacific tomcod	0.9	2.4	2.5	2.1
Pile perch	1	3.3	3.8	< 0.5
Prickly sculpin	< 0.5	< 0.5	1.3	< 0.5
Rock sole	2.7	0.9	< 0.5	< 0.5
Roughback sculpin	0.8	< 0.5	< 0.5	< 0.5
Sand sole	< 0.5	0.8	< 0.5	< 0.5
Shiner surfperch	5.1	19.7	34	103
Snake prickleback	1.3	< 0.5	0.9	< 0.5
Starry flounder	1.1	7.8	10.3	72
Striped perch	< 0.5	0.8	< 0.5	< 0.5
Surf smelt	< 0.5	2	< 0.5	< 0.5
Unknown sculpin	0.8	< 0.5	< 0.5	< 0.5

LDW – Lower Duwamish Waterway
RM – river mile

Anadromous Salmonids—Pacific Salmon

Five species of Pacific salmon (i.e., coho, chinook, chum, sockeye, and pink) have been identified in the LDW study area (Kerwin and Nelson 2000). These anadromous fish use the estuary for rearing and as a migration corridor for adults and juveniles. Among the numerous beneficial uses identified for the LDW study area, use as habitat for outmigrating juvenile salmonids was listed as the most important (Harper-Owes 1983).

Salmonid residence time in the LDW study area is limited and species-specific. Juvenile chinook and chum salmon have been shown to be present from several days to up to 2 months within the LDW, whereas coho salmon pass through the LDW in a few days (Grette and Salo 1986). Sockeye salmon are rare in the LDW. Salmon found in the LDW spawn mainly in the middle reaches of the Green River and its tributaries (Grette and Salo 1986). The juvenile outmigration of all five species generally commences during the high-flow months of March to June. Outmigration usually lasts through mid-July to early August (Nelson et al. 2004; Warner and Fritz 1995). During these months, salmonids use the estuary to feed and begin their physiological adaptation to higher salinity waters.

Of the salmonid species, chinook salmon have been studied the most extensively in the Green-Duwamish system. Puget Sound chinook salmon were listed as threatened under the federal Endangered Species Act (ESA) on March 24, 1999. The decline of chinook salmon has been attributed primarily to habitat degradation and fragmentation, blockage of migratory corridors, impact from hatchery fish, and commercial and local harvesting practices (Myers et al. 1998a).

Other Salmonids

The coastal Puget Sound bull trout population was listed as threatened under the federal ESA on November 1, 1999. Bull trout were historically found in the LDW because it originally included a much larger upper elevation drainage area, including the White River, which still maintains a bull trout run. Current stock status in the Green-Duwamish system is unknown (WDFW 2000). The decline of bull trout has been primarily attributed to habitat degradation and fragmentation, blockage of migratory corridors, poor water quality, historical fisheries practices, and the introduction of non-native species (64 FR 210: 58910-58933).

Bull trout typically spawn in the upper headwaters of a river system, generally requiring high oxygen concentrations and water colder than 10 °C. Although sub-adult bull trout have been captured in the LDW (Shannon 2001; Warner and Fritz 1995), there is no evidence that bull trout are spawned or reared within the LDW.

Both winter- and summer-run steelhead trout, federally listed threatened species, have been observed in the Duwamish/Green River system (Washington Department of Fisheries et al. 1993; WDFW 2007b). Although they were not target species for fish collection, steelhead have been caught during studies conducted as part of the RI.

Summer steelhead (*Oncorhynchus mykiss*) is a non-native stock sustained by wild production of hatchery origin fish (Washington Department of Fisheries et al. 1993). The run size is unknown but approximated to be a few hundred fish. Winter steelhead consist of both wild and hatchery fish with annual returns ranging from 944 to 2,378 fish. Grette and Salo (1986) reported that repeat spawners make up approximately 19% or fewer of returning wild adults in the Green River system. Both winter and summer runs migrate quickly through the LDW (Grette and Salo 1986).

Sea-run cutthroat trout exist in the LDW, but very little is known about this population or its use of the system. Only 11 cutthroat trout were captured in beach seines at nine stations sampled approximately 30 times each throughout the LDW from February through June 1994 (Warner and Fritz 1995).

Non-Salmonid Fish

Based upon the LDW fish studies reviewed in the ERA, the most abundant non-salmonid fish in the LDW are shiner surfperch, snake prickleback, starry flounder, English sole, Pacific staghorn sculpin, Pacific sandlance, juvenile Pacific tomcod, pile perch, rock sole, surf smelt, three-spine stickleback, Pacific herring and longfin smelt (see Appendix A, Attachment A2). In 2006, King County also found tubesnout to be relatively abundant in the downstream portion of the LDW (Anchor and King County 2007). Shiner surfperch, longfin smelt, and Pacific herring are seasonally abundant in the LDW. Pacific sandlance and longfin smelt, though known to be abundant in the LDW, were encountered infrequently in recent beach seine and trawling efforts, as were Pacific herring and surf smelt. Though these species were not encountered frequently during recent sampling, they are occasionally found in large numbers in the LDW (Shannon 2006; Windward 2005b, 2006c). In all previous studies, Pacific staghorn sculpin were consistently one of the most abundant fish captured in the LDW, particularly during the fall months. Pacific staghorn sculpin are opportunistic feeders, and the adults feed at a higher trophic level than other common resident fish species.

Shiner surfperch abundance in the LDW peaks in summer during spawning (Miller et al. 1975). English sole are common in the LDW over all seasons, with peak abundance in spring (Miller et al. 1977b). Adult English sole migrate seasonally to their spawning grounds in Puget Sound in winter (Forrester 1969) and typically spawn in Puget Sound during February and March (Smith 1936). In central Puget Sound, adult populations of English sole spawn in Elliott Bay and Port Gardner but disperse quickly after spawning (Pallson 2001). Starry flounder are also noted to migrate seasonally between very shallow water and in estuaries during the summer, moving into deeper waters in the winter (Morrow 1980).

2.8.4 Wildlife

The aquatic and semi-aquatic habitats of the LDW support a diversity of wildlife species. Formal studies, field observations, and anecdotal reports indicate that up to 87 species of birds and 6 species of mammals use the LDW at least part of the year to feed, rest, or reproduce. This section provides a brief overview of these bird and mammal species. Additional information, including summaries of what is known about how these species use LDW habitats, is provided in Appendix A, Section A.2.2.4.

2.8.4.1 Birds

Many birds have been identified in the LDW area including: passerine/upland birds, raptors, shorebirds/waders, waterfowl, and seabirds; the species are listed in Appendix A (Table A.2-9). Canning et al. (1979) conducted extensive surveys of Kellogg Island, as well as occasional surveys of the entire LDW, from September 1977 to July 1978. They recorded a total of 70 species, including 26 passerines/upland birds, 3 raptors, 11 shorebirds/waders, 17 waterfowl, and 13 seabirds. The study showed that Kellogg Island had a much greater diversity of birds than did the rest of the LDW study area because of its seclusion and greater variety of habitats. Cordell et al. (2001) reported 75 species of birds, including 32 passerine/upland birds, 7 raptors, 8 shorebirds/waders, 16 waterfowl, and 12 seabirds. Diversity and abundance were highest at the Kellogg Island site, but other areas of the LDW were also consistently used by a wide variety of birds. Birds were most abundant in the spring and least abundant in the summer. During the sandpiper presence and habitat survey conducted for the RI in 2004 over the period of four days, 21 bird species were observed while assessing preferred spotted sandpiper habitat areas (Windward 2004j).

Passerines/Upland Birds

Thirty-two species of passerine/upland birds have been documented along the LDW (Canning et al. 1979; Cordell et al. 1999) (see Appendix A, Table A.2-9). Although generally associated with upland habitats, these birds occasionally forage in the exposed mudflats or use intertidal habitats along the LDW for bathing (Canning et al. 1979). During the sandpiper presence and habitat survey, 10 passerine species (starlings; crows; violet green, barn, and tree swallows; house, song, and white-crowned sparrows; rock pigeons, and American robins) were frequently observed in the LDW area (Windward 2004j).

Raptors

Eight species of raptors have been reported to use the LDW: bald eagle, osprey, peregrine falcon, merlin, and four species of hawks (Cordell et al. 1999). During the sandpiper presence and habitat survey bald eagles were frequently observed in the LDW area (Windward 2004j). The bald eagle is listed under ESA as a threatened species but is currently under review for delisting. In Washington, the bald eagle is also listed as a state threatened species (WDFW 2006a, b, 2007a). Five bald eagle nests were occupied within 8 km (5 mi) of the LDW study area in 1999 (King County 1999e). One or two pairs of resident eagles can be found in the LDW vicinity during the summer (King County 1999e), and overwintering migrant eagles are routinely observed from the beginning of October through late March.

In 2006, WDFW reported 10 osprey nest sites located along the LDW or nearby, with nests also observed on Harbor Island and along the Duwamish and Green Rivers within about 6.4 km (4 mi) of the Upper Turning Basin (Thompson 2006). Four osprey nest platforms were identified throughout the LDW during the sandpiper presence and habitat survey, with young observed in each nest (Windward 2004j). Adult ospreys were also frequently observed on tall structures near the nest platforms during this survey.

Two pairs of peregrine falcons currently nest along the LDW. Two peregrine falcons were observed in the LDW area during the sandpiper presence and habitat survey (Windward 2004j). The peregrine falcon is listed as a species of concern under ESA and was downlisted from a state endangered species to a state sensitive species in April 2002 (WDFW 2007a).

Cooper's hawks and sharp-shinned hawks have been observed overwintering in the vicinity of the LDW. Red-tailed hawks are a resident species and are commonly observed along grassland/woodland margins of the LDW. These species are not heavily reliant on the LDW for food; they prey on songbirds and rodents but will also eat waterfowl. Swainson's hawks and merlin are rare in the LDW area (Canning et al. 1979; Cordell et al. 1999).

Shorebirds/Waders

Ten species of shorebirds and wading birds have been documented in the LDW: great blue heron, green heron, three species of sandpipers, dowitcher, dunlin, killdeer, sanderling, and lesser yellowlegs (Cordell et al. 1999). Of these species, great blue heron are year-round residents and comprise the only sizeable and consistent population recorded by Cordell et al. (1996) on the Duwamish River. During the sandpiper presence and habitat survey, great blue heron were frequently observed throughout the LDW (Windward 2004j). One nesting great blue heron colony is located 11 km (6.8 mi) to the northwest of the LDW (Kiwanis Ravine colony); another colony is located 12 km (7.5 mi) to the southeast, in Renton (Black River colony) (Butler 1997). A former blue heron colony of up to 37 nests was located a few hundred meters to the west of Kellogg Island. The colony was likely abandoned in 2000 because of local disturbances or predation (Norman 2006).

The two most common shorebirds observed in the LDW are killdeer and various species of sandpipers. The spotted, least, and western sandpipers are reported to use the LDW, and spotted sandpipers have been observed nesting on Kellogg Island (Canning et al. 1979). Spotted sandpipers are present in the Northwest region year-round; whereas least and Western sandpipers are present during the winter

(Sibley 2003). During the sandpiper presence and habitat survey, small numbers of spotted sandpipers were observed in the LDW area, particularly on and in the vicinity of Kellogg Island and Herring's House Park (Windward 2004j). Killdeer were also observed during the survey in the same areas as spotted sandpiper, but in larger numbers. Though no spotted sandpiper nests were observed during the survey, a killdeer nest was observed on Kellogg Island. Least and western sandpipers occur in mixed flocks and are difficult to distinguish. These species nest primarily in northern Canada and Alaska in the summer months but are reported to frequent Kellogg Island from September through May. Most are thought to be migrants, though some may reside in the LDW area throughout the winter.

Waterfowl

Seventeen species of waterfowl, including 13 species of ducks, 3 species of geese, and the American coot, use the LDW (Cordell et al. 1999). During the sandpiper presence and habitat survey, Canada geese and mallard ducks were commonly observed, and gadwall and common mergansers were also observed though less frequently (Windward 2004j). In general, these migratory birds overwinter in the Puget Sound area (and further south) and migrate north in the summer. A small resident population of mallards lives year-round in the LDW area, and as many as 290 migratory mallards have been reported to move through the LDW.

Canvasback, greater scaup, gadwalls, bufflehead, and both common and Barrow's goldeneye are reported to use the LDW. Feeding by all diving duck species is centered around Kellogg Island (Canning et al. 1979).

All three species of North American mergansers use the LDW study area. Approximately 30 red-breasted mergansers reportedly overwinter in the LDW area between December and March.

A population of approximately 1,000 Canada geese resides in the vicinity of Lake Washington and includes the LDW population (Canning et al. 1979).

Seabirds

Fourteen species of seabirds were recorded in the LDW during surveys completed in 1979 and 1999 (Canning et al. 1979; Cordell et al. 1997; 1999), including two species of cormorants, three species of grebes, three species of gulls, three species of loons, pigeon guillemot, common murre, and Caspian tern. Wintering cormorants (pelagic and double-crested) use the LDW study area between November and May, with large numbers present between December and April. During the sandpiper presence and habitat survey, double-crested cormorants and gull species were commonly observed in the LDW area (Windward 2004j).

Glaucous-winged gulls and mew gulls are the only gull species reported to use the LDW study area in large numbers. Glaucous-winged gulls are reported to use the area year-round; mew gulls frequent the area from September through May (Canning et al.

1979). Caspian terns have been seen on Kellogg Island (Luxon 2004). Pigeon guillemots and common murres have also been reported in the LDW; however, their use of the LDW is infrequent. The common loon is a state sensitive species; they are reported to be rare in the LDW area (Canning et al. 1979). Three species of grebe have been reported in the LDW area. The Western Grebe is present in the Northwest region during the winter (Sibley 2003). Grebes and other marine bird species have been declining throughout the greater Puget Sound region in recent years (Nysewander et al. 2001).

2.8.4.2 Mammals

Three species of terrestrial mammals use the LDW (i.e., raccoons, muskrats, and river otters), and three marine mammal species may enter the LDW (i.e., harbor seal, California sea lion, and harbor porpoise) (Dexter et al. 1981; Tanner 1991; WDFW 1999).

Anecdotal information indicates that a river otter family lives year-round on Kellogg Island in the LDW, although otters have not been observed during wildlife surveys (Cordell et al. 2001). Raccoons are reported to be common along the forested ridge slopes to the west of the LDW. Muskrat populations are reported to exist at T-107 and the Upper Turning Basin (Canning et al. 1979).

Harbor seals and sea lions are commonly seen in Elliott Bay and have occasionally been observed in the LDW. During a survey conducted by WDFW from December 1998 to June 1999, sea lions were observed in the LDW on 16 occasions and seals on 17 occasions (WDFW 1999), with most observations of both species occurring below the 1st Avenue S Bridge. Harbor seals have been shown to forage over large distances ranging from 5 km (3.1 mi) (Stewart et al. 1989) to 55 km (34.2 mi) (Beach et al. 1985). Recent information on use of the LDW by harbor porpoises was not available, although it has been noted that they occasionally enter the LDW (Dexter et al. 1981).

2.8.5 Plants

Few studies have investigated the plant communities present in the LDW (Canning et al. 1979; Cordell et al. 2001; Tanner 1991; USFWS 2000a). The methods used to assess plant communities have ranged from analysis of aerial photos to field surveys. Most recently, Cordell et al. (2001) monitored the vegetation of wetland restoration and reference sites in the LDW by conducting surveys during the growing season at six restoration and six reference sites throughout the LDW from 1993 to 1999.

In the LDW, there is a total of 0.0175 km^2 (0.0068 mi^2) of habitat with estuarine macrophytes, primarily limited to portions of Kellogg Island and other small intertidal areas with vegetated intertidal habitat (USFWS 2000b, c). *Carex* (sedges) and *Scirpus* (bulrushes) are the predominant marsh vegetation types between the Upper Turning Basin and Kellogg Island. Downstream from Kellogg Island are more marine plants such as *Salicornia* (grassworts), *Distichlis* (salt grass), and *Atriplex* (salt bush). The interior high marsh plant community of Kellogg Island, which is flooded only by higher spring tides, includes *Carex lyngbyei*, *Distichlis spicata*, *Juncus balticus* (Baltic rush), and *Phragmites* sp., a non-native species (Battelle et al. 2001). The naturally occurring *Carex* patches surveyed in 1993 occurred between elevations of 1.6 and 3.0 m (5.2 and 9.8 ft) above MLLW, and the single patch of naturally occurring *Scirpus* was at 3.7 m (12 ft) above MLLW, indicating that these plants are seldom underwater (Cordell et al. 2001).

2.8.6 Threatened and endangered species

Fifteen species identified in the LDW study area are listed under either the ESA or by WDFW, as candidate species, threatened species, or species of concern (see Table 2-12). Information on the presence and observed frequency of these species in the LDW area is provided in the ERA (Appendix A).

Table 2-12. LDW species listed under ESA or by the Washington State Department of Fish and Wildlife

COMMON NAME	SCIENTIFIC NAME	STATUS
Chinook salmon	*Oncorhynchus tshawytscha*	federal threatened species, state candidate species
Coho salmon	*Oncorhynchus kisutch*	federal candidate species
Puget Sound steelhead	*Oncorhynchus mykiss*	federal threatened species
River lamprey	*Lampetra ayresi*	federal species of concern, state candidate species
Bull trout	*Salvelinus confluentes*	federal threatened species, state candidate species
Pacific herring	*Clupea herengus pallasi*	federal candidate species, state candidate species
Pacific cod	*Gadus macrocephalus*	federal species of concern, state candidate species
Walleye pollock	*Theragra chalcogrammus*	federal candidate species, state candidate species
Rockfish species	*Sebastes* spp.	state candidate species
Bald eagle	*Haliaeetus leucocephalus*	federal threatened species,[a] state threatened species
Peregrine falcon	*Falco peregrinus*	federal species of concern, state sensitive species[b]
Merlin	*Falco columbarius*	state candidate species
Common murre	*Uria aalge*	state candidate species
Common loon	*Gavia immer*	state sensitive species
Western grebe	*Aechmophorus occidentalis*	state candidate species

Source: (WDFW 2007a)

Note: The Puget Sound southern resident orca distinct population segment is listed on both the federal and Washington State endangered species lists. Orca do not use the LDW but are occasionally present in Elliott Bay and may be exposed to chemicals from the LDW through the consumption of prey (such as chum salmon) that spend part of their time in the LDW.

[a] Listing currently under review for removal.

[b] Downlisted from state endangered to state sensitive on April 2002 (WDFW 2002).

ESA – Endangered Species Act

LDW – Lower Duwamish Waterway

2.9 HUMAN USE

This section provides a brief overview of the demography, land use, and site use in the LDW area.

2.9.1 Demography

Although the area surrounding the LDW is often regarded as an industrial corridor, residential neighborhoods (i.e., South Park and Georgetown) are present in the LDW area. The racial diversity of the residential population in the immediate vicinity of the LDW is greater than the racial diversity of residents within the City of Seattle overall or the nearby City of Tukwila (Table 2-13).

Table 2-13. Population data for Duwamish neighborhoods, City of Seattle, and City of Tukwila

RACE[a]	POPULATION		
	DUWAMISH NEIGHBORHOODS[b]	CITY OF SEATTLE[c]	CITY OF TUKWILA[d]
Caucasian	11,871 (53.5%)	394,889 (70.1%)	10,032 (58.3%)
African-American	2,143 (9.7%)	47,541 (8.4%)	2,310 (13.5%)
Native American and Alaska Native	397 (1.8%)	5,659 (1.0%)	196 (1.1%)
Asian	4,221 (19.0%)	73,910 (13.1%)	1,788 (10.5%)
Native Hawaiian and other Pacific Islander	333 (1.5%)	2,804 (0.5%)	332 (1.9%)
Other single race	1,868 (8.4%)	13,423 (2.4%)	1,348 (7.8%)
Two or more races	1,359 (6.1%)	25,148 (4.5%)	1,198 (6.9%)
Total	**22,192 (100%)**	**563,374 (100%)**	**17,204 (100%)**

Note: Percentage of total population is given in parentheses.

[a] The US Census Bureau does not consider Hispanic or Latino to be a race but rather an ethnicity (2001). Thus, individuals who identify themselves as being Hispanic or Latino may be of any race. As a result, the Hispanic/Latino population was not included in the total population or percentage calculations as a separate group. Approximately 3,039 people in the Duwamish neighborhoods, 29,719 people in the City of Seattle, and 2,353 people in the City of Tukwila identified themselves as being of Hispanic or Latino origin.

[b] Population estimates from Seattle LDW neighborhoods, including Tract 93, Duwamish/SODO; Tract 99, North Delridge; Tract 108, Riverview; Tract 109, Georgetown; Tract 112, South Park; and Tracts 113, 264, and 265, Highland Park (City of Seattle 2006).

[c] Data from Census 2000 (City of Seattle 2006).

[d] Data from Washington State Office of Financial Management (2002).

2.9.2 Land use and ownership

The LDW study area is part of WRIA 9 (Map 2-2). WRIA 9 extends from the headwaters of the Green River in the Cascade Mountains to the mouth of the Duwamish River at Elliott Bay. Map 2-12 presents current land ownership of properties adjacent to the LDW. Tables 2-14 and 2-15 provide information on land ownership and land use, respectively, within WRIA 9. Over 77% of the land in WRIA 9 is under private ownership, with approximately 8% owned by the federal government and 8% owned by the State of Washington. Approximately 6% of land in

WRIA 9 is owned by local jurisdictions, and a very small percentage of the river basin is owned by tribal entities.

Table 2-14. Land ownership in the Duwamish and Green River Basin (WRIA 9)

OWNER	NO. OF ACRES	PROPORTION
Federal	30,634	8.2%
State	29,512	8.0%
Local	23,980	6.4%
Tribal	319	0.1%
Private	287,911	77.3%
Total	**372,356**	**100.0%**

Source: Ecology (2006c)

Table 2-15. Land use in the Duwamish and Green River Basin (WRIA 9)

LAND USE CATEGORY	PROPORTION
Forested	54%
Rangeland	5%
Agricultural	5%
Urban	18%
Barren	10%
Open water	8%
Total	**100.0%**

Source: Ecology (2006c)

Land use within the greater Duwamish drainage basin has changed considerably since the construction of Harbor Island and the LDW nearly 100 years ago. Table 2-16 identifies the changes in land use in the lower 11 miles of the Duwamish River watershed since that time.

Table 2-16. Historical and current land cover from RM 0.0 to RM 11.0

LAND COVER TYPE	HISTORICAL COVERAGE (ac)	CURRENT COVERAGE (ac)
Forested floodplain	3,385	51.9
Forest terrace	885	na
Shrub	na	27.2
Upland bedrock	72	na
Bare earth and mudflats	na	39.5
Herbaceous vegetation	1.5	157.9
Wetlands	3,252	51.4
Built/developed	0	6,315.9
Open water	699.8	766.5

LAND COVER TYPE	HISTORICAL COVERAGE (ac)	CURRENT COVERAGE (ac)
Lakes/ponds	12.4	0.5
Mainstream river channel	687.4	766

Source: King County (2005k)
ac – acres
na – not available
RM – river mile

The LDW shoreline is zoned predominantly for recreation, conservancy preservation, and urban industrial land use (ATSDR 2003). A small area on the west side of the LDW between RM 3.1 and RM 3.3 is zoned for single-family residential use (SPU 2007). A shoreline survey of the LDW was completed for the WRIA 9 Steering Committee and SPU in 2004 (Terralogic and Landau 2004). The survey assessed shoreline conditions from RM 6.0 to the northern tip of Harbor Island (including both the East and West Waterway shoreline), and survey results are presented in Table 2-17.

Table 2-17. Results of the shoreline survey conducted for the LDW and the East and West Waterways

DESIGNATION	% OF SHORELINE SURVEYED	TOTAL MILES
Shoreline/Bank Structures		
Shoreline armoring – bulkheads	31.93%	7.04
Shoreline armoring – riprap	56.05%	12.36
Concrete boat ramps	0.68%	0.15
Unarmored bank area	11.34%	2.50
Total	**100%**	**22.05**

Source: WRIA 9 Subcommittee and SPU (Terralogic and Landau 2004)
Note: The Lower Duwamish study area for the shoreline survey was defined as the LDW from RM 0.0 to RM 6.0 and the East and West Waterways to the northern tip of Harbor Island. Therefore, this study area includes more than the LDW Superfund site, which extends only to the southern tip of Harbor Island.
LDW – Lower Duwamish Waterway
RM – river mile

Over 85% of the shoreline in the surveyed area was armored (with either bulkheads or riprap). Just over 10% of the shoreline was not armored, and a very small portion (less than 1%) was covered by concrete boat ramps.

2.9.3 Site use

Approximately 43% of the Duwamish estuary subwatershed (extending from RM 11.0 to Elliott Bay [Map 2-2]) is used for commercial/industrial purposes, and approximately 39% is residential (King County 2005k).

2.9.3.1 Commercial and industrial site use

The LDW serves as a navigational corridor for the movement of raw materials and products used by local industries. Many of the industrial and commercial facilities on the LDW operate year-round. In 2005, the City of Seattle studied the industrial land zoning and usage in Seattle (City of Seattle 2005). This study identified 3,980 ac of industrial land surrounding the LDW, which equals approximately 77% of the land zoned for industrial use in Seattle. According to the study (City of Seattle 2005), the four types of zoning within the Duwamish industrial area are Industrial General 1 (IG1), Industrial General 2 (IG2), Industrial Buffer (IB), and Industrial Commercial (IC). IG1 (with 54% of the 3,980 ac) allows for the heaviest of manufacturing. IG2 (42%) is similar to IG1 but allows for more office and retail development. IB (3%) allows for light and general manufacturing and more office and retail development. IC (1%) allows for the most office development.

2.9.3.2 Tribal use

The LDW is frequently used by tribes as a resource and for cultural purposes. The Muckleshoot Indian Tribe and Suquamish Tribe are both federally recognized tribes and are natural resource trustees for the Duwamish River. The Muckleshoot Indian Tribe currently conducts seasonal commercial, ceremonial, and subsistence netfishing operations in the LDW. The Suquamish Tribe actively manages resources north (downstream) of the Spokane Street Bridge, located just north of the LDW.

Other tribal activities include the Duwamish Tribe's use of Herring's House Park to launch canoes and for various cultural events (Windward 2005j). The Duwamish Tribe also uses T-107 for monthly gatherings, children's dance group practices, and large annual cultural gatherings. The Duwamish Tribe has recently completed construction of a longhouse and cultural center near T-107 and Herring's House Park, which will likely increase their use of these two areas along the LDW.

2.9.3.3 Recreational site use

The LDW is not a major area for recreational use compared with other water bodies in and around the City of Seattle (King County 1999e). However, several public access points from which people may enter the LDW for recreational purposes exist. In the human access survey conducted along the LDW shoreline in 2005 (Windward 2005j), 93 commercial/industrial, residential, and public properties along the LDW were surveyed for human access and use potential. Of these 93 properties, 27 sites were identified as potential human access sites: 7 on the eastern shore of the LDW, 18 on the western shore, and 2 at the southern end of Harbor Island (see Map B.3-1). Most of these access points were on King County or City of Seattle public access grounds or located at the ends of streets. Many of the access points were elevated above the water surface and separated from the sediment by steep banks covered by riprap or blackberry bushes.

Street end parks and restoration sites such as Gateway Park, Herring's House Park, Duwamish River Park, and the Duwamish Diagonal Way access point provide easy access to the LDW shoreline and waterway (NOAA 2007b; DRCC and ECOSS 2008; Windward 2005j). Many of these access points serve as boat launches and sites for recreational activity on the LDW. In addition, residents living adjacent to the waterway have easy access to the LDW. Residents use the shore in front of their or their neighbors' property to launch boats or as a site for bird watching, swimming, walking, or picnicking (Windward 2005j). Public kayaking tours of the LDW are also offered on a regular basis by Alki Kayak Tours (2008).

Three marinas are located in the LDW study area. The human access survey (Windward 2005j) identified 17 additional locations used for the launching or pullout of hand-powered boats or motorboats. In addition, 8 sites along the LDW have been used for swimming, and 10 have been used for picnicking. The frequency of recreational activities at these sites may increase in the future as ongoing remedial efforts and habitat restoration projects are completed, but such activities are likely to remain limited as a result of the active industrial and commercial use of the LDW and the availability of other, nearby areas that provide superior recreational opportunities.

Fishing, crabbing, and shellfishing activities have been reported at 34 sites along the LDW shoreline, and individuals have also been observed fishing off the sides of barges moored in the LDW (Windward 2005j). The access survey (Windward 2005j) and a clam abundance survey (Windward 2004c) also discussed the potential for fishing and clamming activities to occur throughout the LDW study area (see Map B.3-2). However, the clam survey did not assess areas farther upstream than RM 3.5; the potential for clamming (and the relative abundance of clams) is currently unknown upstream of this point (Windward 2004c). Through treaty rights, tribal members have access to both public and private property and can harvest clams and other shellfish along the entire shoreline of the LDW (Windward 2007b). In addition, the navigable waters of the LDW are generally owned by the Port of Seattle, as successor to the King County Commercial Waterway District, and are accessible to the public by boat.

Several surveys have documented relatively high seafood consumption among several Asian and Pacific Islander (EPA 1999a) and Native American (Suquamish Tribe 2000; Toy et al. 1996) populations within the Puget Sound region, some of whom may fish within the LDW for recreational or subsistence purposes. A review of seafood consumption surveys for the Elliott Bay/Duwamish and larger Puget Sound area was conducted by ESG (1999). The review indicated that the potentially exposed population of fish consumers in the LDW study area consists of a diverse group of people that fish both from shore and boat. In 1997, a fishing survey was also completed by King County (1999c). This intensive survey was conducted for 2 months in the summer of 1997, the period of highest use, and showed that use in the LDW was significantly lower than in other Puget Sound sites. However, people fish seasonally for salmon in the LDW, and a relatively high number of individuals fished/crabbed from the Spokane Street Bridge, just north of the LDW. Specifically, seafood has been harvested from T-105 within the

LDW and from the Spokane Street Bridge just north of the LDW adjacent to Harbor Island. Seafood consumption rates are discussed in greater detail in the baseline HHRA (Appendix B).

The Washington State Department of Health (WSDOH) issued a seafood consumption advisory for the LDW in July 2002, recommending that people limit their consumption of resident fish and shellfish from the LDW. In 2005, WSDOH updated the advisory with the recommendation that no resident fish or shellfish be consumed from the LDW (WSDOH 2005), and this advisory is still in effect. WDFW is responsible for educating anglers and others who harvest seafood from the LDW of the advisory; however, WDFW does not enforce the advisory with citations or penalties (Windward 2007b). WSDOH and Public Health – Seattle & King County also recommend that resident fish and shellfish from any urban shorelines along Elliott Bay should not be consumed; commercial shellfish harvesting is restricted because of high counts of fecal coliform bacteria (King County 1999d; WSDOH 2008a).

2.9.3.4 Residential site use

Two mixed residential/commercial neighborhoods are located adjacent to the LDW (see Appendix B, Map B.1-1). The South Park neighborhood, within and adjacent to the southern edge of the Seattle city limit, borders the west bank of the LDW and encompasses approximately 300 m (984 ft) of residential shoreline (Green-Duwamish Watershed Alliance 1998). The Georgetown neighborhood, located east of the LDW and E Marginal Way S, is separated from the LDW by several commercial facilities, although access to the LDW by foot from this neighborhood is possible. Based on data from the 2000 census, the combined residential population of the South Park and Georgetown neighborhoods is 4,900 (US Census Bureau 2002; ATSDR 2003). During the 15 years prior to 2003, the population of the Georgetown neighborhood decreased slightly, while the population of the South Park neighborhood increased by over 30% (ATSDR 2003).

2.9.3.5 Restoration activities

At least 12 habitat restoration projects are ongoing along the LDW, with several more planned for the near future (Windward 2005j; NOAA 2007a; EPA 2006e; WRIA 9 Steering Committee 2002). These activities include efforts to restore salmon habitat and efforts to develop street end parks and improve recreational access. Restoration efforts are expected to continue over time. Examples of restoration projects that include human use or public access components along the waterway include Herring's House Park, T-107, T-105, three sites on the southwest side of the Upper Turning Basin, including the Coastal America restoration site, the Diagonal Avenue S/T-108 public shoreline access area, Cecil B. Moses Park, and the 8th Avenue S street end park/Gateway North restoration area (DRCC and ECOSS 2008). Map 2-8 provides the locations of these restoration areas and street end parks. Map 2-8 also provides the locations of restoration areas that do not include human use or public access.

2.10 SUMMARY

The LDW study area consists of the downstream portion of the Duwamish River, excluding the East and West Waterways. The LDW is a well-stratified estuary that has been extensively modified over the last 100 years.

The LDW serves as a major shipping route for bulk and containerized cargo, and the shoreline along the majority of the LDW has been developed for industrial and commercial operations. Common shoreline features include constructed bulkheads, piers, wharves, sheet piling walls, buildings that extend over the water, and steeply sloped banks armored with riprap or other fill material.

To facilitate navigation and economic development, the LDW was straightened and has been frequently dredged in many areas. The excavated material was used to fill the old channel areas and the lowlands to bring them above flood levels. Subsequent filling of the lowlands for continued development resulted in a surficial layer of fill over most of the lower Duwamish Valley. Most of the upland areas adjacent to the LDW have been heavily industrialized for many decades.

Throughout the 1900s, the watershed area and flow volumes to the Duwamish River were reduced by about 70% as a result of the diversion of the river's tributaries. Today, the Green River is the primary source of water for the Duwamish River. Surface water runoff within the LDW area also contributes to flow within the LDW, from sources such as SDs, tributary creeks, CSOs, and non-point inputs; these sources are expected to be less than 1% of the total discharge, even during peak flow events. Annual average downstream flow for the Duwamish River, measured at the Auburn gaging station, ranged from 4.3 to 329 m^3/s (152 to 11,600 cfs [the record high]) between 1962 and 2004.

Water circulation within the LDW is driven by tidal actions and river flow. Fresher water moving downstream overlies the tidally influenced saltwater entering the system. Typical of tidally influenced estuaries, the LDW has a relatively sharp interface between the low-salinity water outflow at the surface and saltwater inflow at depth.

Tidal action in the LDW greatly affects groundwater flow direction, flow rate, and overall water quality. Under typical conditions, the hydraulic gradient is toward the LDW. At low tide, the hydraulic gradient between the groundwater system and the LDW is typically at its highest, causing the flow of local groundwater into the LDW. At high tide, the hydraulic gradient can reverse. Tidal action causes a continuous oscillation of groundwater within the effective zone of influence. The periodic reversal in flow direction caused by tidal influences can decrease the migration of dissolved chemicals in groundwater.

Sediment composition varies greatly throughout the LDW, ranging from sands to mud (fine-grained silt and clays), depending on the source of the sediments and the local current velocity. Sediments in the navigation channel near the Upper Turning Basin are predominantly sands, whereas sediments toward the mouth of the river are predominantly fine-grained silts.

Ecological habitats of the LDW have been modified extensively since the late 1800s through hydraulic changes, channel dredging, the filling of surrounding floodplains, and the construction of overwater structures, levees, dikes, and other bank stabilization structures. The remnants of natural meanders along the LDW, several of which are now used as slips, and the area west of Kellogg Island are the only evidence of the river's original winding course. Small intertidal areas of marsh and unvegetated marsh habitat in the LDW have become the focus of habitat restoration activities.

Benthic invertebrate communities observed in the LDW consisted of 670 taxa, representing 178 families in 13 phyla. Typical of estuarine environments, the benthic invertebrate community was dominated by annelid worms, mollusks, and crustaceans. The most abundant large epibenthic invertebrates are crabs and shrimp. Based on SPI information, 10 cm is an appropriate estimate of the BAZ in the LDW. The LDW is inhabited by numerous anadromous and resident fish species. Fifty-three resident and non-resident fish species were captured in the LDW during sampling events conducted for the RI. The aquatic and semi-aquatic habitats of the LDW support a diversity of wildlife species. Formal studies, field observations, and anecdotal reports indicate that up to 87 species of birds and 6 species of mammals use the LDW at least part of the year to feed, rest, or reproduce (i.e., raccoons, muskrats, and river otters, harbor seal, California sea lion, and harbor porpoise). Fifteen species identified in the LDW study area are listed under either the ESA or by WDFW, as candidate species, threatened species, or species of concern (see Table 2-12).

Approximately 43% of the Duwamish estuary subwatershed is used for commercial/ industrial purposes, and approximately 39% is residential. Two residential neighborhoods, South Park and Georgetown, are located adjacent to the LDW. The Muckleshoot Tribe currently conducts seasonal commercial, ceremonial, and subsistence netfishing operations on the LDW, which is part of their U&A fishing grounds; and the Suquamish Tribe actively manages resources that depend on the LDW for at least part of their life cycle, north (downstream) of the Spokane Street Bridge, located just north of the LDW. The Duwamish Tribe also uses several areas for gatherings and cultural events.

The LDW is not a major area for recreational use compared with other water bodies in and around Seattle. However, several public access points from which people may enter the LDW for recreational purposes exist. Although recreational use may increase at some point in the future, this area is anticipated to remain primarily commercial, industrial, and residential in use.

In 2005, WSDOH recommended that no resident fish or shellfish be consumed from the LDW (WSDOH 2005); this advisory is still in effect.

3 Sediment Dynamics

The Phase 1 RI (Windward 2003a) summarized existing information on sediment dynamics in the LDW and yielded a basic understanding of the stability of bedded sediments and sediment transport in the system, which was used to develop a preliminary conceptual site model (CSM). This CSM was considered to be preliminary because sufficient site-specific information and data were not available during the Phase 1 RI to confirm it. Based on the data available at the time of the Phase 1 RI, the preliminary CSM postulated:

- The LDW is net depositional[23] on a site-wide scale.
- On a local scale, the sediment bed is either aggrading (i.e., sediment bed elevation is increasing as a result of sediment deposition) or in dynamic equilibrium[24] (i.e., sediment bed elevation is neither increasing nor decreasing).
- Bed erosion occurs only episodically and over small spatial scales.

As described in the Phase 2 work plan (Windward 2004h) and the sediment transport data report (Windward and QEA 2005), additional data were collected in 2004 and 2005, consistent with EPA (2005b) guidance, to support development of a concise description of LDW sediment transport processes that affect sediment stability in the system and to refine the Phase 1 CSM. These additional data and supporting analyses are summarized in Section 3.1 and presented in detail in the sediment transport analysis report (STAR) (Windward and QEA 2008).

Based on information presented in the STAR (Windward and QEA 2008), LDWG, EPA, and Ecology concluded that additional sediment transport modeling (STM) was warranted because some of the key questions about sediment dynamics could not be satisfactorily answered by the empirical approach described in Section 3.1. This modeling has been completed and is summarized in Section 3.2 and presented in detail in the STM report (QEA 2008).

The analyses conducted during the STM, in conjunction with empirical and ship-induced bed scour analyses from the STAR, have resulted in an improved

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[23] The term net depositional refers to a condition in which a portion of the sediment bed, or a reach of the river or waterway, experiences more deposition (i.e., settling of sediment from the water column onto the bed) than erosion (i.e., scour from the bed to the water column) over periods of about 1 year or longer (i.e., annual time scales). The net sedimentation rate is the rate at which net deposition occurs.

[24] The term dynamic equilibrium refers to a condition in which the sediment bed is neither net erosional nor net depositional, with minimal changes in bed elevation occurring over annual time scales. The bed may experience episodic erosion as a result of high-flow events or ship-induced bed scour, or net deposition over short time scales.

understanding of sediment transport processes in the LDW. Results and findings from the major components of these analyses were integrated and synthesized to produce a clear and concise picture of sediment transport over the scale of the entire LDW, as summarized in Section 3.3. This information was also used to update the CSM, as summarized in Section 3.4, and will be used in conjunction with site-specific characteristics (e.g., berthing areas, presence of outfalls) to assess the effectiveness of various remedial alternatives in the FS.

3.1 EVALUATION OF SEDIMENT DYNAMICS

Several types of empirical data were evaluated in the STAR (Windward and QEA 2008) to assess sediment dynamics using a weight-of-evidence approach. Those data are discussed in the sections that follow:

- Sediment physical properties (e.g., sediment particle size) (Section 3.1.1)
- Sediment cores used for age dating (Section 3.1.2)
- Bathymetry (Section 3.1.3)
- Physical and chemical markers in sediment cores for estimating net sedimentation rates (Section 3.1.4)
- Sediment cores used to evaluate erosion properties (Section 3.1.5)

The sediment cores discussed in Sections 3.1.2 and 3.1.5 were collected specifically for the evaluation of sediment dynamics in the RI; the data described in the other sections were collected for other purposes. These data were used in modeling analyses that are described in Sections 3.1.6 (hydrodynamic model) and 3.1.7 (ship-induced bed scour model). These modeling analyses are not the same as the STM application discussed in Section 3.2.

3.1.1 Sediment physical properties

The physical characteristics of bedded sediment affect sediment transport processes that occur in a given area. A large amount of historical data exists on the physical properties of sediment in the LDW (i.e., sediment particle size, TOC). Additional physical property data were derived from the sediment cores collected as part of the STAR (Windward and QEA 2008).

In the STAR (Windward and QEA 2008), historical surface sediment particle size data, which were available from approximately 1,250 locations sampled since 1990, were evaluated for spatial trends. The data were variable in the bench areas and navigation channel, with no apparent spatial structure (e.g., downstream fining). Sediment in the navigation channel had a higher silt+clay content than did other sediment, with an average silt+clay content of 62% (10th and 90th percentiles were 29 and 82%, respectively). Silt+clay content was more variable outside of the navigation channel, including the Upper Turning Basin, with 10th and 90th percentiles of 13 and 87%, respectively, and an average value of 53%. The area above RM 5.0 had a relatively low average silt+clay content (approximately 11.5%).

Additional sediment particle size data were collected in December 2004 as part of the STAR (Windward and QEA 2008). These data were generated using a laser particle size counter, in contrast to the sieve and pipette method used to generate the historical data. The laser particle size counter generates an almost continuous particle size distribution, using approximately 100 different particle size categories, rather than the 15 particle size categories generated from the sieve and pipette method. Consequently, the data from the 19 sediment cores collected in December 2004 are discussed separately from the historical data because different summary statistics can be generated from the much larger number of particle size categories.

The particle size distributions in the top 5 cm of the Sedflume[25] cores collected at 19 locations in the LDW (Map 3-1) are summarized in Table 3-1. Eighteen cores were classified as sandy silt, and one core (Sf-5) was classified as silty sand (Windward and QEA 2005, 2006). A statistical analysis of the 18 sandy silt cores indicated that the median particle diameter (D_{50}) averaged 30 μm, and the clay, silt, and sand contents averaged 6%, 69%, and 28%, respectively (Table 3-1). In general, variability in particle size in these cores was relatively low. For example, about 90% of the D_{50} values for individual samples varied between 10 and 50 μm (silt). The single core classified as silty sand (Sf-5) had a D_{50} of 329 μm and a sand content of 79%.

Table 3-1. Particle size data for surface layer (0 to 5 cm) of Sedflume cores

SEDIMENT CORE ID	D_{10} (μm)[a]	D_{50} (μm)[a]	D_{90} (μm)[a]	CLAY CONTENT (%)	SILT CONTENT (%)	SAND CONTENT (%)
Sf-1	2	12	60	9	82	9
Sf-2	3	24	115	6	72	22
Sf-3	3	19	117	7	75	18
Sf-4	3	19	223	7	71	22
Sf-5	11	329	739	2	19	79
Sf-6-R1	3	21	122	7	73	20
Sf-6-R2	4	39	184	5	60	35
Sf-7	4	41	1,112	5	55	40
Sf-8	3	28	235	6	68	26
Sf-9	5	79	402	4	43	53
Sf-10	3	23	89	6	78	16
Sf-11	4	94	653	5	41	54

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[25] The particle size data discussed in this section were collected as supporting data from cores tested in a device called a Sedflume, which was used to measure gross erosion rates over a range of shear stresses, as discussed in greater detail in Section 3.1.5.

SEDIMENT CORE ID	D_{10} (µm)[a]	D_{50} (µm)[a]	D_{90} (µm)[a]	CLAY CONTENT (%)	SILT CONTENT (%)	SAND CONTENT (%)
Sf-12	4	24	139	5	73	22
Sf-13	3	21	97	7	76	17
Sf-14	3	22	161	7	70	23
Sf-15	2	16	73	8	81	11
Sf-16-R1	3	20	93	7	76	17
Sf-16-R2	2	18	82	8	78	14
Sf-17	3	19	88	7	78	15
Average[b]	3	30	225	6	69	25
Standard deviation[b]	0.8	22	265	1.3	12	13
95% confidence interval[b]	2 – 4	19 – 41	93 – 357	5 – 7	63 – 75	17 – 31
Minimum[b]	2	12	60	4	43	9
Maximum[b]	5	94	112	9	82	54

Source: Windward and QEA (2005, 2006)

[a] D_{10}, D_{50}, and D_{90} values are the 10th, 50th (median), and 90th percentiles of the particle diameter data for that core.

[b] Statistics exclude data for Sf-5, which was classified as silty sand; the other 18 cores were classified as sandy silt.

ID – identification

In general, the 18 sandy silt cores had more silt and less sand than surface sediment samples contained in the surface sediment dataset, which averaged approximately 41% silt and 41% sand. These differences reflect one of the specific objectives of the Sedflume data collection, which was to target bench areas expected to be depositional. The LDW-wide surface sediment grain size dataset is much more comprehensive and larger than the Sedflume dataset and reflects many different types of sediments in the LDW.

Wet density data do not exist in the historical baseline surface sediment dataset, but this parameter was analyzed in the Sedflume cores. Wet densities of the 18 sandy silt cores (excluding Sf-5) ranged from about 1.2 to 1.7 g/cm^3. The single silty sand core (Sf-5) had a higher wet density (1.8 to 1.9 g/cm^3). In general, wet density increased with depth within a core because of consolidation effects. Similar to the particle size distributions, wet densities in the Sedflume cores were relatively uniform throughout the LDW for a given depth. A correlation analysis was conducted between wet density and particle size. The analysis indicated that minimal correlation existed between these two bulk bed properties for the 18 sandy silt cores. For example, the correlation coefficient (r) between wet density and D_{50} values for the 18 sandy silt cores was 0.42.

TOC content has been analyzed in over 1,300 surface sediment samples collected throughout the LDW since 1990. In general, TOC content in surface sediment did not vary widely. Outside the navigation channel, the 10th and 90th percentiles were 0.80 and 2.9%, respectively. The TOC content in the navigation channel was less variable than the TOC content outside the navigation channel, with 10th and 90th percentiles of 1.2 and 2.6%, respectively. The average TOC content (1.9%) was the same within and outside the navigation channel. The area above RM 5.0 had a lower average TOC content (0.84%).

3.1.2 Geochronology analysis

To facilitate a better understanding of how net sedimentation rates in bench areas compared with rates previously estimated for the navigation channel, a geochronology study was conducted as part of the STAR (Windward and QEA 2008). This study consisted of an age-dating analysis of sediment cores collected from the LDW in December 2004 using the radioisotopes cesium-137 (^{137}Cs) and lead-210 (^{210}Pb).

Fourteen sediment cores were collected from bench areas within the LDW in 2005 (Map 3-2). Each core was segmented in 1-cm intervals, and every fifth segment within the top 81 cm (i.e., 0 to 1 cm, 5 to 6 cm, etc., to 80 to 81 cm) was submitted for ^{137}Cs, ^{210}Pb, TOC, and total solids content analyses; remaining segments were archived. After the initial results were reviewed, additional intervals, some as deep as 111 cm, were analyzed in four cores (Sg-3, Sg-6, Sg-7, and Sg-13) in an attempt to detect a ^{137}Cs peak at a depth greater than 81 cm.

Estimated net sedimentation rates based on the geochronology analysis are presented in Table 3-2. A detailed explanation of the calculation methods is provided in Windward and QEA (2008). Direct comparisons of the net sedimentation rates generated from the two different radioisotopes were possible for four cores: Sg-2, Sg-5a, Sg-7, and Sg-10. For two of the four cores (Sg-2 and Sg-5a), the ^{137}Cs net sedimentation rate fell within the range of the ^{210}Pb net sedimentation rate. In the other two cores (Sg-7 and Sg-10), net sedimentation rates determined from the ^{210}Pb data were lower than those estimated using the ^{137}Cs data. The fact that sedimentation rates could not be estimated for some cores suggests the possibility of localized disturbances.

Table 3-2. Comparison of net sedimentation rates from ^{137}Cs and ^{210}Pb analyses

Sediment Core ID	Mud Line Elevation (ft MLLW)	Range of Estimated Net Sedimentation Rates from ^{137}Cs Analysis (cm/yr)		Range of Estimated Net Sedimentation Rates from ^{210}Pb Analysis (cm/yr)
		Via ^{137}Cs Peak	Via First Presence of Detectable ^{137}Cs	
Sg-1a	-17.7	0.9 – 1.1	1.2 – 1.3	ne
Sg-2	-3.5	ne	0.5 – 0.6	0.4 – 1.1
Sg-3	-21.4	1.9 – 2.1	nd	ne
Sg-4	-12.8	1.6 – 2.0	nd	ne
Sg-5a[a]	-16.5	1.4 – 1.6	nd	0.7 – 9.3
Sg-6	-17.9	2.5 – 2.7	nd	ne

Sediment Core ID	Mud Line Elevation (ft MLLW)	Range of Estimated Net Sedimentation Rates from ^{137}Cs Analysis (cm/yr)		Range of Estimated Net Sedimentation Rates from ^{210}Pb Analysis (cm/yr)
		Via ^{137}Cs Peak	Via First Presence of Detectable ^{137}Cs	
Sg-7[a]	-16.1	1.9 – 2.1	nd	0.5 – 1.1
Sg-8	-10.3	ne	ne	ne
Sg-9	+0.4	0.3 – 0.5	0.8 – 0.9	ne
Sg-10	-11.0	1.6 – 1.8	nd	0.2 – 1.0
Sg-11c	-1.3	ne	ne	ne
Sg-11b	+0.6	ne	0.6 – 0.7	ne
Sg-12	-11.1	> 2.0	nd	ne
Sg-13	-8.3	2.3 – 2.6	nd	ne

[a] These cores were located relatively close to maintenance dredging areas in the 1990s. Consequently, the uncertainty surrounding the estimated net sedimentation rates for these cores is greater than the uncertainty surrounding estimated net sedimentation rates for cores not located near historical dredged areas.

Cs – cesium
ID – identification
MLLW – mean lower low water
nd – not detected
ne – not estimated
Pb – lead
yr – year

Net sedimentation rates in the bench areas were estimated to range from 0.2 to > 2.0 cm/yr. In general, net sedimentation rates were greater than 1 cm/year in the subtidal bench areas and less than 1 cm/year in the intertidal areas. Thus, net sedimentation rates in the subtidal bench and intertidal areas were lower than those previously estimated for the LDW navigation channel (i.e., 1 to 15 cm/year between RM 0.0 and RM 1.7, 10 to 25 cm/year between RM 1.7 and RM 3.4, and 20 to 110 cm/year between RM 3.4 and RM 4.7) (Windward 2003a). No other consistent spatial trend was apparent in the estimated net sedimentation rates calculated from radioisotope age-dating methods, likely because the data were too sparse to develop spatial relationships.

Several physical, chemical, and biological factors introduce uncertainty into the net sedimentation rates estimated from the radioisotope profiles. These factors include: 1) natural variability in the radioisotope data, 2) variations in sediment physical characteristics, 3) physical disturbances of the sediments (e.g., erosion, historical dredging), 4) compression and/or mixing of sediments during core collection/extrusion, and 5) poor sediment recovery rates in core samples. The first three factors are likely the greatest contributors to uncertainty in net sedimentation rates estimated during the geochronology study, primarily because of uncertainty in the extent and magnitude to which these factors vary in the LDW.

The geochronology analysis was one of two independent empirical lines of evidence used to derive net sedimentation rate data, the other being an analysis of multiple time markers in subsurface sediment cores (see Section 3.1.4). Because estimates based on time markers were available for many locations, the uncertainties discussed above for the geochronology analysis do not greatly affect the LDW-wide interpretation of net sedimentation characteristics. Consequently, the net sedimentation rate was selected as the model calibration target, as described in Section 3.2.3.

3.1.3 Bathymetric analysis

The results of bathymetric surveys conducted in different years were also compared in the STAR (Windward and QEA 2008) to evaluate sediment dynamics. Sediment bed (i.e., mud line) elevations at specific locations were compared to identify areas of potential scour or deposition. A similar analysis was conducted for the Phase 1 RI (Windward 2003a), but additional bathymetric data had been collected since that time.

The bathymetric analysis conducted for the STAR consisted of a comparison of two existing bathymetric datasets collected from a portion of the LDW (between RM 2.6 and RM 4.7) in 2000 and from the entire LDW in 2003, resulting in a qualitative assessment of potential scour and depositional areas. Sediment surface elevations in 2000 were defined using a combination of bathymetric measurements from surveys conducted for USACE[26] and Boeing (Pentec et al. 2001). Sediment surface elevations in 2003 were defined using the acoustic multi-beam bathymetric data collected by David Evans and Associates (Windward and DEA 2004). Because of the higher density of measurements in 2003 (relative to 2000), each measurement from 2000 was paired with the nearest measurement from 2003 and a change in sediment elevation at each paired location was computed. The maximum distance allowed between pairs was 10 ft. Approximately 49,000 paired locations were evaluated. Linear interpolation was performed between measurement locations to estimate areas of scour and deposition. The largest differences measured (approximately 3 ft) represent potential scour (i.e., more recent measurements up to 3 ft lower than older measurements), deposition (i.e., more recent measurements up to 3 ft higher than older measurements), or a combination of both.

There were several uncertainties associated with positioning and measurement accuracy for the bathymetric surveys (Windward 2003a). The imprecision associated with these uncertainties has the greatest potential effect on estimates for the steep sloped areas of the LDW or at the outermost limits of the survey transects. These uncertainties are further compounded by the fact that the three surveys were performed by different contractors, using different equipment and different methods. Moreover, sediment elevation measurements from the USACE and Boeing surveys in 2000, which were used to define year 2000 conditions in the LDW, were collected along transects spaced at approximately 200-ft intervals, making it difficult to find matching points in the 2003 survey. Based on the uncertainties summarized above, any change in elevation less than 12 in. (30 cm) at a particular point was not considered significant.

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[26] Bathymetry charts as contained in USACE file number E-12-2.1-12.3, dated September 15, 2000.

Measurement inaccuracies and other uncertainties described above made it difficult to make reliable conclusions regarding fine-scale sediment transport based on this analysis. Nonetheless, very small areas of scour and deposition in excess of 12 in. (30 cm) were noted in these comparisons. However, most of these areas were located immediately adjacent to the navigation channel (which has been routinely dredged), in the Upper Turning Basin, or at the outermost limits of the survey data; these areas are likely associated with measurement inaccuracies that are in excess of the criterion used in this analysis (i.e., 12 in. [30 cm]).

3.1.4 Additional analysis of net sedimentation rates

In the STAR (Windward and QEA 2008), net sedimentation rates were estimated from the geochronology cores, as described in Section 3.1.2, and evaluated along with several independent chemical and physical lines of evidence including historical information and sediment chemistry cores collected in 2006. The purpose of these additional analyses was to estimate net sedimentation rates for areas other than those where geochronology cores were collected and to validate the net sedimentation rates calculated from the geochronology study.

The site data used in the additional analyses included chemistry and stratigraphy data from 56 subsurface sediment cores collected in 2006 and from historical subsurface sediment cores; grain size distribution data; dredging records; chemical spill, industrial, and regional discharge records; and bathymetric data. These data provided a set of time markers that were found at different depths in the sediment bed at various locations in the LDW. After assigning a date or time period and specific depth for a particular marker, the net sedimentation rate was estimated from that marker. This estimate represented the average rate of net deposition for the time period between the time marker and core collection.

Nine different physical and chemical time markers were evaluated, although not all markers could be evaluated for a given core (Table 3-3). The conclusion of this time marker analysis was that net sedimentation rates were spatially variable, with the highest rates in the navigation channel (greater than 2 cm/yr), moderate rates in the subtidal bench areas (less than 2 cm/yr), and the lowest rates in the intertidal bench areas (less than 0.5 cm/yr). These results are generalized trends; there are several specific net sedimentation rate estimates greater than 2 cm/year outside of the navigation channel, as shown on Map 3-3. Net sedimentation rates were not calculated for the Duwamish/Diagonal, Slip 4, Boeing Plant 2/Jorgensen Forge, and Terminal 117 (T-117) EAAs because those areas are being evaluated separately by parties responsible for those removal actions. Net sedimentation rates calculated by the STM, in conjunction with other lines of evidence (e.g., empirical data), will be incorporated into the natural recovery analyses, which could potentially include the EAAs, to be conducted in the FS.

There are some uncertainties associated with the analysis of physical and chemical time markers, and the estimated net sedimentation rates may be underestimated or overestimated as a result of the following factors:

- Other, unaccounted-for physical events or disturbances
- Differences between measured and in situ core depths
- Variability in net sedimentation rates estimated among different time markers in the same core
- Lag time associated with the chemical or physical age dates
- Lack of spatial coverage in some areas

Additional details on how each marker was evaluated are provided in Attachment F of the STAR (Windward and QEA 2008), which was prepared by RETEC (now AECOM).

Although geochronology cores, as summarized in Section 3.1.2, were not co-located with the sediment cores for which the empirical sedimentation rates were estimated, the geochronology-based estimates generally agreed with the rates estimated from the physical and chemical markers, particularly at locations where the two core locations were close together. Because of the consistency in the findings of the analysis of geochronology cores and the evaluation of other empirical lines of evidence, it was possible to combine the net sedimentation rates estimated from the two methods to develop a large-scale view of LDW net sedimentation, including a map showing the spatial distribution of estimated net sedimentation rates in the LDW (Map 3-3). This map depicts five categories of net sedimentation rates: < 0.5 cm/yr, 0.5 to 1.0 cm/yr, 1.0 to 1.5 cm/yr, 1.5 to 2.0 cm/yr, and > 2.0 cm/year (the specific rate estimates for each core are also provided). Results of the analysis provided a consistent picture of large-scale net sedimentation processes in the LDW.

Table 3-3. Net sedimentation rates in the LDW estimated from physical and chemical time markers

Subsurface Core ID	Approx. River Mile	Estimated Net Sedimentation Rates (cm/yr) Determined from Time Markers and Event Horizons[a]								
		Physical[b]			Chemical (1-ft, 2-ft Intervals)			Chemical (6-in. Intervals) from a Subset of 2006 Cores		
		Lower (Native) Alluvium[c]	Hanson Dam Const.[d]	Dredge Horizon[e]	Lead/ PCB/ Phthalate Introduction	PCB Peak Usage/ Spill	Source Control Efforts	PCB Introduction	PCB Peak Usage/ Spill	Source Control Efforts
		1916	1961	Variable	1920/1935/ 1950	1960/1974	1980	1935	1960/1974	1980
RI 2006 Cores										
SC-1	0.0	0.9			1.7			1.1	0.9	0.9
SC-2	**0.1**									
SC-3	0.1	0.4								
SC-4	0.2	1.1			1.7		1.2			
SC-5	0.2	0.7	0.5		0.5, 0.9					
SC-6	0.3	2.6	3.0		2.3		2.3	2.6	2.3	2.7
SC-7	0.3				0.7					
SC-8	0.4					3.3	1.2			
SC-9	0.5		1.8	1.5						
SC-10	0.5	2.4	2.7	2.4		2.9				
SC-11	0.5		0.5		0.4					
SC-12	0.6	2.3	1.8		2.9	2.0	2.3, 1.2	2.6	2.0	2.1
SC-13	0.9				1.1			1.1 to 2.1		
SC-14	0.9	2.9			4.0					
SC-15	0.9	2.5	1.4		3.0	4.8				
SC-16	0.9	2.4			3.8	2.9	2.3, 1.2			
SC-17	1.0					2.9				
SC-18	1.0	1.9		1.5, 1.9	0.7, 0.9, 1.1					
SC-19	1.0	3.0	4.7		3.4	4.3				
SC-20	**1.0**									
SC-21	1.0	3.3	3.4		2.7	4.9	2.3			
SC-22	**1.1**									

Table 3-3, cont. Net sedimentation rates in the LDW estimated from physical and chemical time markers

Subsurface Core ID	Approx. River Mile	Estimated Net Sedimentation Rates (cm/yr) Determined from Time Markers and Event Horizons[a]								
		Physical[b]			Chemical (1-ft, 2-ft Intervals)			Chemical (6-in. Intervals) from a Subset of 2006 Cores		
		Lower (Native) Alluvium[c]	Hanson Dam Const.[d]	Dredge Horizon[e]	Lead/ PCB/ Phthalate Introduction	PCB Peak Usage/ Spill	Source Control Efforts	PCB Introduction	PCB Peak Usage/ Spill	Source Control Efforts
		1916	1961	Variable	1920/1935/ 1950	1960/1974	1980	1935	1960/1974	1980
SC-23	1.3		3.3		4.3	4.8	4.7	3.4	4.8	3.3
SC-24	1.2	1.1	0.7		0.7, 0.9					
SC-25	1.3	2.0 to 2.5			2.5, 3.0		1.2			
SC-26	**1.4**									
SC-27	1.4	1.5 to 2.6						1.4	1.2	0.9
SC-28	**1.4**									
SC-29	1.4	0.6	0.4							
SC-30	1.6	1.1								
SC-31	1.7			12.2	1.0, 1.2, 1.5					
SC-32	1.7	1.7 to 2.4	1.9		2.0, 2.5					
SC-33	1.9	2.9			3.0, 3.8			2.6	0.8 to 1.7	0.9, 1.4
SC-34	1.9		2.2							
SC-35	2.0		3.5	2.8, 3.7						
SC-36	2.1	2.8	2.2							
SC-37	2.1	1.8	1.8		2.0, 2.6	1.0	2.3			
SC-38	**2.1**									
SC-39	2.2	2.9								
SC-40	2.2	0.7								
SC-41	2.4	2.6								
SC-42	2.5		2.7							
SC-43	2.6	3.0	0.5							
SC-44	2.7				1.4, 1.1			1.3	0.5	0.3
SC-45	**2.8**									
SC-46	2.7	2.3		7.6, 1.8						

Table 3-3, cont. Net sedimentation rates in the LDW estimated from physical and chemical time markers

	Estimated Net Sedimentation Rates (cm/yr) Determined from Time Markers and Event Horizons[a]									
		Physical[b]			Chemical (1-ft, 2-ft Intervals)			Chemical (6-in. Intervals) from a Subset of 2006 Cores		
		Lower (Native) Alluvium[c]	Hanson Dam Const.[d]	Dredge Horizon[e]	Lead/ PCB/ Phthalate Introduction	PCB Peak Usage/ Spill	Source Control Efforts	PCB Introduction	PCB Peak Usage/ Spill	Source Control Efforts
Subsurface Core ID	Approx. River Mile	1916	1961	Variable	1920/1935/ 1950	1960/1974	1980	1935	1960/1974	1980
SC-47	3.1	1.0			1.3, 1.4, 2.2	1.0	1.2			
SC-48	3.3				0.4 to 0.5					
SC-49	3.5		2.4		4.3					
SC-50	3.8	0.9			1.0, 1.2, 1.5					
SC-51	3.8									0.6
SC-52	3.9				0.5, 0.7, 0.9					
SC-53	4.2	3.1	3.3							
SC-54	4.3	1.8	2.7							
SC-55	4.9	1.0	0.3							
SC-56	4.7				0.8 to 1.0					
Historical Cores										
B3 (T105 1985)	0.2	4.9	5.1							
DUD006 (D/D 1994)	0.5					1.9, 3.1	2.7			
DR18 (PSDDA99)	1.8	2.2	3.2							
DR39 (PSDDA99)	2.2	1.5								
SC11 (Slip 4 2004)	2.8	1.5	2.2							
S3 (PSDDA98)	3.8	3.0	3.3							
HI-NR-01	< 0						1.4			
HI-NR-02	< 0						1.3			
HI-NR-03	< 0						2.5			
HI-NR-04	< 0						1.3			

Source: Analysis conducted by RETEC (now AECOM), as reported in Windward and QEA (2008), Appendix F.

Notes: See subsurface data report (Windward and RETEC 2007) for core logs. Ranges or multiple estimates are given in some table cells where chemical markers for different chemicals could be observed.

[a] All net sedimentation rate estimates are based on recovered core depths and do not include uncertainty.

Table 3-3, cont. Net sedimentation rates in the LDW estimated from physical and chemical time markers

[b] Sediments were grouped into three stratigraphic units identified for the LDW, primarily based on density, color, sediment type, texture, and marker bed horizons. The three sediment stratigraphy units were identified as follows: Recent, Upper Alluvium, Lower (Native) Alluvium.

[c] Lower (Native) Alluvium is defined by top of dense sand unit.

[d] Hanson Dam construction is defined by the presence of organic silt.

[e] Dredge event rates show rate for event to top of core and intra-marker rate from stratigraphic marker to dredge effects marker.

ID – identification

LDW – Lower Duwamish Waterway

PCB – polychlorinated biphenyl

Blank cells indicate that markers were not present or core was not clearly indicative of a strong time marker.

Bold text, gray shading indicate no strong marker; therefore, no calculation was made for the core.

3.1.5 Site-specific erosion properties

Erosion rates as a function of shear stress and depth in the sediment bed were evaluated using 19 sediment cores collected from 17 locations in the LDW (Map 3-1) in December 2004 as part of the STAR (Windward and QEA 2004, 2005). Fourteen of these cores were collected from bench areas, and five cores were collected from the navigation channel. Many of the cores collected from the bench areas for erosion property analysis were located relatively close to the geochronology cores discussed in Section 3.1.2. Duplicate cores were collected at two locations (Sf-6 and Sf-16), one from a bench area and one from the navigation channel.

Erosion rates as a function of depth in the bed and shear stress were measured in the laboratory over the top 30 cm of each core using a Sedflume (Windward and QEA 2004), which has proven useful for measuring the erosion properties of sediments at many other sites (McNeil et al. 1996). The principal purpose of the Sedflume is to measure gross erosion rates (E_{gross}) over a range of applied shear stresses (τ)[27] at various depths in a sediment core.

Erosion rate data, collected from at least 15 different aquatic systems, clearly show that a log-linear relationship exists between gross erosion rate (E_{gross}) and bed shear stress (τ). Equation 3-1 portrays this relationship as a power-law equation (2000):

$$E_{gross} = A\,\tau^{n} \text{ when } \tau > \tau_{cr} \qquad \textbf{Equation 3-1}$$

$$E_{gross} = 0 \text{ when } \tau \leq \tau_{cr}$$

Where:

E_{gross} = gross erosion rate (cm/s)

A = site-specific erosion rate parameter that may be dependent on the sediment dry density (ρ), as well as on other sediment bed properties (units vary, as discussed below)

τ = bottom shear stress (Pa)

n = site-specific erosion rate exponent (unitless)

τ_{cr} = critical shear stress (Pa)

The conversion of Equation 3-1 to log space shows the log-linear characteristics of this equation.

$$\log(E_{gross}) = n \log(\tau + \log(A) \qquad \textbf{Equation 3-2}$$

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[27] Shear stress is the force that acts parallel to a plane. In this application, the force is hydrodynamic (from moving water) and the plane is the sediment bed surface.

To determine the values of A and *n*, a log-linear correlation analysis was conducted using the erosion rate (i.e., Sedflume) data for a specific core (or from a particular interval within a core), where E_{gross} was plotted as a function of τ. The units of the coefficient A vary, depending on the value of the exponent *n*.

The critical shear stress (τ_{cr}), defined as the shear stress at which a small but measurable rate of erosion is observed, was estimated from Sedflume erosion rate data. For Sedflume studies performed at other sites (Jepsen et al. 2001; Jones 2000; McNeil et al. 1996), the critical erosion rate was set at 10^{-4} cm/s, a value that consistently corresponded to the initiation of erosion. Thus, a critical erosion rate of 10^{-4} cm/s was used in this study (Windward and QEA 2008). The critical shear stress was calculated by rearranging Equation 3-1:

$$\tau_{cr} = \left(\frac{E_{gross}}{A} \right)^{\frac{1}{n}}$$

Equation 3-3

where E_{gross} was set equal to 10^{-4} cm/s.

The data generated from the Sedflume analysis was used to assess the spatial variability (i.e., horizontal and vertical) of the erosion rate parameters (A, *n*, and τ_{cr}). The equations given above for these erosion rate parameters do not explicitly account for physical mixing processes such as bioturbation. Sediment-dwelling invertebrates (e.g., clams, worms, amphipods) can alter sediment properties, which may affect the erosion and transport of surface sediments. However, the effects of physical mixing processes are implicitly included in the calculation of erosion rate parameters because the characteristics of the Sedflume cores reflect the net effect of all physical processes that have occurred in the past at the specific locations where the cores were collected. Seasonal differences in biological activity may exist within the sediment layers evaluated in the Sedflume device (i.e., top 30 cm). The Sedflume cores were collected in December; many studies of benthic invertebrate communities were conducted in the spring, summer, or fall.

SPI data were collected from within the LDW in the summer of 2006 (Ecology 2007n) (Section 2.8.2.2). The SPI study documented the widespread presence of Stage 3 communities, which represent larger, longer-lived organisms that typically burrow and feed deeper in the sediment column and occur in relatively stable, undisturbed habitats. It is likely that these communities would be present in the LDW regardless of the season, because of their longevity. Many organisms in benthic invertebrate communities are sedentary and construct tubes or mucus-lined burrows; these structures along with biological activity increase the roughness of the sediment surface. Increased surface roughness tends to decrease flow velocities at the sediment-water interface (i.e., boundary layer) by introducing turbulence; mucus acts as a binding agent. Given the results of the SPI survey, the Sedflume data collected in December are considered representative of the effects from benthic invertebrates over a large spatial scale within the LDW, even though seasonal differences in boundary roughness and erosion have not been characterized.

The erosion rate relationship (i.e., Equation 3-1) was similar among various cores within a specific 5-cm depth layer (Windward and QEA 2008), as presented in Table 3-4. This similarity suggested that 5-cm subsections of cores could be separated into different groups for a given depth layer, as defined by similarities in the erosion rate parameters A and n. This grouping was conducted in an attempt to evaluate whether LDW-wide generalizations could be made using these data. Sixteen core groups were created, each consisting of two to nine core segments. Upon visual inspection, two core segments were considered to be outliers and were not included in any core group.

The results for the individual cores within a group were combined to determine average erosion parameter values for that group. The average exponent (n) value for a group was the arithmetic average of the n values for the cores within the group. The average proportionality constant (A_{ave}) was determined by calculating the log-average value:

$$\log(A_{ave}) = (1/K) \sum \log(A_k) \qquad \textbf{Equation 3-4}$$

where K was the total number of cores in the group. Equation 3-2, which is based on standard statistical theory for linear equations, was used to determine the average erosion rate parameters for a group of core intervals. This approach resulted in calculations of the arithmetic average of the exponent n values and the log-average of the A values (see Equation 3-4) to determine the appropriate average erosion rate parameters for the group.

The vertical distribution of τ_{cr} values for all of the core groups is shown in Figure 3-1. In general, critical shear stress increased with depth in the sediment bed, as expected because of bed consolidation. It would take more energy or higher water flows to erode sediments at depth compared with unconsolidated surface sediments. The group-averaged A, n, and τ_{cr} values determined from this analysis were used in developing the horizontal and vertical spatial distributions of these parameters for input to the LDW STM (Section 3.2).

Table 3-4. Values for erosion rate parameter (A) and erosion rate exponent (*n*) for each Sedflume core interval

SEDIMENT CORE ID	0-TO-5-cm INTERVAL		5-TO-10-cm INTERVAL		10-TO-15-cm INTERVAL		15-TO-20-cm INTERVAL		20-TO-25-cm INTERVAL	
	A ($\times 10^{-4}$)	*n*	A ($\times 10^{-4}$)	*n*	A ($\times 10^{-4}$)	*n*	A ($\times 10^{-4}$)	*n*	A ($\times 10^{-4}$)	*n*
Sf-1	3.8	3.0	0.29	3.5	5.3	0.3	3.8	3.0	0.13	2.7
Sf-2	8.8	1.7	1.9	2.0	0.19	2.9	9.2	1.3	0.51	3.1
Sf-3	89	2.2	24	3.3	12	2.7	16	2.2	29	1.8
Sf-4	23	2.1	3.7	2.6	0.74	3.3	0.55	3.1	0.03	5.0
Sf-6-R1	24	2.8	3.8	3.1	0.7	3.2	2.9	2.1	1.4	2.5
Sf-6-R2	35	2.2	6.3	2.6	4.3	3.0	0.27	3.7	0.62	3.1
Sf-7	19	1.5	6.0	1.8	11	2.0	1.8	3.0	0.35	3.6
Sf-8	21	1.5	5.9	2.8	8.2	2.1	0.14	3.0	0.023	4.2
Sf-9	15	0.96	0.055	3.4	24	2.3	0.029	3.2	0.0	6.7
Sf-10	22	1.5	5.9	2.8	8.3	3.7	0.083	3.5	0.0083	4.7
Sf-11	19	1.4	3.9	3.1	0.51	3.5	9.4	1.4	0.097	3.2
Sf-12	6.5	3.9	8.3	3.0	6.1	2.1	7.5	3.4	8.0	3.3
Sf-13	11	1.3	0.7	2.9	0.43	5.2	0.095	3.7	0.96	3.1
Sf-14	80	2.3	6.9	1.6	0.35	2.8	1.6	2.2	2.5	1.8
Sf-15	9.4	1.7	6.0	2.5	0.097	3.5	0.88	2.1	240	-0.53
Sf-16-R1	43	2.7	3.3	2.9	15	2.6	5.1	3.5	3.7	1.6
Sf-16-R2	29	2.6	4.7	2.6	5.3	0.3	3.8	3.0	0.016	4.0
Sf-17	21	3.0	25	2.6	0.19	2.9	9.2	1.3	2.8	2.3

ID – identification

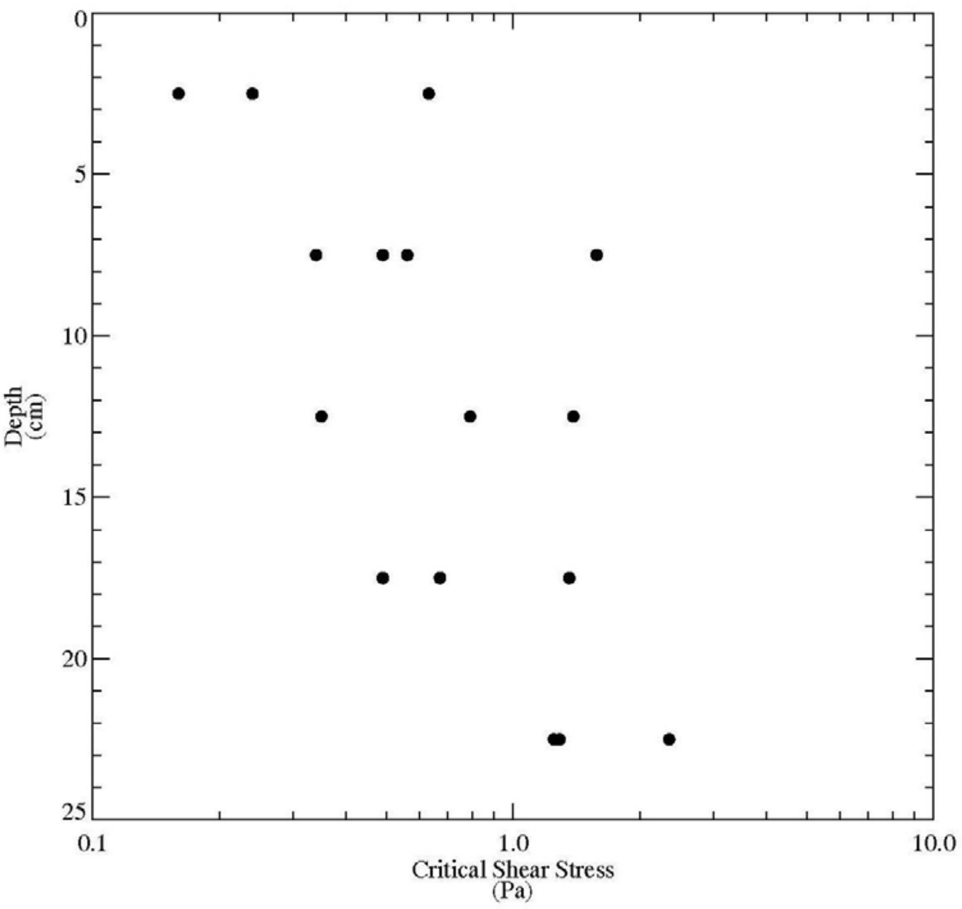

Figure 3-1. Vertical distribution of group-averaged critical shear stress values

3.1.6 Potential effects of natural events

The potential effects of an episodic high-flow event on bed stability in the LDW were evaluated through the use of a hydrodynamic model in the STAR (Windward and QEA 2008). The hydrodynamic model was used to simulate circulation in the LDW for a range of high-flow events (e.g., events with return periods of 2, 10, and 100 yrs) and a range of tidal conditions (i.e., spring and neap tides). Results of the hydrodynamic simulations were used to develop inferences and hypotheses about bed scour in the LDW during high-flow events and the spatial distribution of excess shear stress (i.e., bed shear stress minus critical shear stress).

High-flow events in the LDW are episodic in nature, with typical durations of hours to days. These episodic events may cause bed scour or deposition to occur at specific locations in the LDW. Although it is possible that a particular high-flow event may have measurable effects on bed elevation (i.e., either net deposition or erosion) at some locations, the long-term effects of such an event on the depositional or erosional environment[28] at those locations may be relatively minor.

Sediment transport during an episodic high-flow event in the LDW is affected by a range of factors, including hydrodynamic circulation, spatial and temporal variations in bed shear stress, deposition, erosion (including bed armoring and consolidation effects), and sediment loads from upstream sources. These factors are the primary determinants of the effects of a high-flow event on bed stability and sediment transport in the LDW. Thus, a quantitative evaluation of sediment transport during a high-flow event requires the application of hydrodynamic and sediment transport models that incorporate all of the processes discussed above.

A three-dimensional hydrodynamic model of the LDW and Elliott Bay was developed during a water quality study of Elliott Bay, the East and West Waterways, and the LDW (King County 1999e). This model was created using the Environmental Fluid Dynamics Code (EFDC). Calibration and validation results indicated that the model simulated hydrodynamic processes in the LDW and Elliott Bay with reasonable accuracy. Dr. Earl Hayter of EPA increased the resolution of the King County model by developing a finer grid structure (Arega and Hayter 2004). The modified model was re-calibrated, as presented in Attachment B of the STAR (Windward and QEA 2008).

The LDW is a saltwater wedge estuary, and its hydrography has been studied as part of various investigations (Harper-Owes 1981; Santos and Stoner 1972). The location of the toe of the saltwater wedge varies depending on the rate of freshwater inflow from the Green/Duwamish River and the stage and magnitude of the saltwater tidal flow entering the LDW from Elliott Bay. The location of the saltwater wedge not only affects circulation in the LDW but also has an effect on bed shear stress distributions and bed scour during high-flow events. Results from hydrodynamic simulations of various high-flow events have suggested that the LDW may be broadly separated into three hydrodynamic reaches during high-flow conditions (Windward and QEA 2008).[29]

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[28] Depositional environment refers to an area in which the sediment bed is net depositional (i.e., bed elevation increases) over annual time scales. The bed may experience episodic erosion as a result of high-flow events or ship-induced bed scour. Erosional environment refers to an area in which the sediment bed is net erosional (i.e., bed elevation decreases) over annual time scales. The bed may experience net deposition over time scales of less than a year.

[29] In the STAR, reaches were designated 1 (most downstream), 2 (middle reach), and 3 (most upstream reach), but those designations are not used here so as to avoid confusion with the same reach designations used in Section 3.2 to describe the STM model-derived reaches. The boundaries between the STAR reaches were RM 2.0 (between Reach 1 and Reach 2) and RM 3.0 (between Reach 2 and Reach 3).

Various modeling analyses were conducted in the STAR (Windward and QEA 2008) to evaluate potential bed scour in the LDW during high-flow events (i.e., discharge greater than or equal to the 2-year high-flow event). The results of these analyses are presented in the STAR and summarized below.

- Over all flow conditions, bed shear stress tended to be higher in the navigation channel than in the bench areas.
- During high-flow events in the most downstream reach, negligible bed scour was estimated to occur in most of the area downstream of RM 1.8. Between RM 1.8 and RM 2.0, excess shear stress values of 0.4 pascal (Pa) or less were estimated to occur during high-flow events.
- During high-flow events in the middle and most upstream reaches (i.e., upstream of the saltwater wedge):
    - Minor differences were estimated in the general spatial pattern of excess shear stress during ebb and flood tides. Higher bed shear stresses were estimated during spring tide than during neap tides.
    - In general, lower excess shear stresses were estimated in the bench areas than in the navigation channel for a given high-flow event and tidal condition.
    - Within the portions of the bench areas where erosion was estimated to occur, the potential for erosion tended to be highest near the navigation channel and tended to decrease toward the shoreline.
    - The most upstream reach was estimated to have higher excess shear stress values than the middle reach.
- The hydrodynamic energy regime (represented by excess shear stress) during high-flow events was not estimated to have a significant effect on the erosion rate properties or geochronology at a particular location within the bench areas of the LDW. No correlation was apparent between the structure (i.e., vertical distribution of erosion rate or physical properties) of either a Sedflume core or a geochronology core and the hydrodynamic energy regime at the core location.

Although areas of potential erosion were identified through this analysis, the depth of bed scour in those areas during a specific high-flow event could not be reliably estimated without the application of an STM (see Section 3.2).

3.1.7 Potential effects of ship-induced bed scour

The gross bed scour that could potentially result from ship traffic was also analyzed in the STAR using a numerical model (Windward and QEA 2008). The focus of this analysis was on the upstream and downstream movements of ships within the navigation channel. This analysis was necessarily limited to estimating gross bed scour depth because sediment transport and deposition processes were not explicitly accounted for in the model. Consequently, bed scour results, expressed on a per ship passage basis, were considered to be upper-bound estimates, with the actual bed erosion resulting from ship traffic in the navigation channel likely to be less than that estimated because in a typical event, sediment is suspended for a brief period of time but is then re-deposited shortly thereafter once the propeller-driven forces are no longer present. This analysis was considered to be a screening-level evaluation, with order-of-magnitude accuracy at best. The model was not designed to assess cumulative effects of ship traffic. Furthermore, the model did not address effects from vessel maneuvering outside of the navigation channel, which could result in localized scour and will be considered in the FS. Detailed ship-induced bed scour studies will be required during the design of future sediment cleanup actions. The STM discussed in Section 3.2 describes long-term simulations that incorporate cumulative effects of all ship traffic.

Results of this modeling analysis included:

- Within the navigation channel, ship movement was estimated to result in an average bed scour depth of less than 1 cm per ship passage in the most downstream reach, and less than 0.1 cm per ship passage in the middle and upstream reaches. Within the bench areas, average bed scour depths of about 1 to 2 cm per ship passage were estimated in the most downstream reach, and less than 1 cm per ship passage was estimated in the middle and upstream reaches.
- A sensitivity analysis indicated that reducing ship speed from the LDW speed limit of 5 knots to a speed of 2.5 knots would likely significantly reduce bed scour, with estimated bed scour of less than 1 cm throughout the LDW for all conditions. Doubling the applied ship power (i.e., horsepower) had minimal effect on estimated scour depth.
- The reworked (i.e., mixed) sediment layer was equated with the depth of gross bed scour, based on the assumption that the same layer is continually reworked. The most downstream reach was estimated to have an upper-bound average scour thickness of less than about 1 cm in the navigation channel and about 1 to 2 cm in bench areas. In the middle and upstream reaches, the reworked sediment layer was estimated to have an upper-bound average thickness of less than 0.1 cm in the navigation channel and less than 1 cm in bench areas. The frequency of mixing is about 100 to 250 events per year.
- Ship-induced bed scour depths were not found to be significantly related to erosion rate properties or geochronology results at any particular location in the bench areas or navigation channel of the LDW. No correlation was apparent between the structure of a geochronology or Sedflume core and the amount of ship-induced bed scour at the core location.

- The present-day structure of the LDW sediment bed reflects the effects of ship-induced bed scour that has occurred for at least 40 years.[30] The general lack of evidence in the bathymetric record of large depressions in areas with frequent ship traffic validates the assumption that the upper sediment layer is continually reworked and suggests that ship activity is not a major cause of sediment transport in the LDW. In some areas, ships may have caused localized erosion from physical forces unrelated to propeller-driven scour (i.e., anchors dragging), but the model did not account for such activities.

3.2 SEDIMENT TRANSPORT MODELING

The analyses presented in the STAR (Windward and QEA 2008) produced a significant amount of information on LDW sediment characteristics. However, a limitation of those analyses is the inability to predict erosion, deposition, and net sedimentation throughout the LDW during high-flow events and over multi-year periods. It was recognized by LDWG, Ecology, and EPA that development of an STM could potentially enhance the efficacy of various analyses during the FS process, including monitored natural recovery (MNR) analyses.

Discussions and meetings among LDWG, Ecology, and EPA during July and August 2006 concerning an STM resulted in the formation of an STM group,[31] which worked collaboratively and provided advice on the development, calibration, and application of the model. The STM group held meetings, conference calls, and informal discussions among members at various times between August 2006 and May 2007 to review model status and discuss the next steps to be taken in model development, calibration, and application. This collaborative effort was instrumental in producing a consensus modeling framework and in achieving the study objectives described in Section 3.2.1, which summarizes the results of the STM effort (QEA 2008).

3.2.1 Modeling objectives

The overall objectives of the STM effort were to:

- Develop a quantitative tool to evaluate short- and long-term sediment transport processes in the LDW
- Refine the CSM for the LDW that was developed in the STAR

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[30] Many of the businesses that rely on barge or tug traffic, or their predecessors, have existed for at least 40 years.

[31] The members of this group have included: Shane Cherry (Cherry Creek Environmental), Karl Eriksen (USACE), Joe Gailani (USACE), Earl Hayter (EPA), Brad Helland (Ecology), Bruce Nairn (King County), Mike Riley (S.S. Papadopulos & Associates), Peter Rude (City of Seattle), Beth Schmoyer (City of Seattle), David Schuchardt (City of Seattle), Jeff Stern (King County), Kym Takasaki (USACE), and Kirk Ziegler (QEA).

- Provide information to support FS analyses and inform remedial decision-making

Multi-year simulations were conducted to predict long-term changes in bed elevation (i.e., net sedimentation rate) as well as changes in surface layer sediment composition. These results will be used in the FS to help evaluate remedial alternatives and estimate the rate of natural recovery in the LDW attributable to sediment transport processes. Through the use of long-term (e.g., 30-yr) simulations, the STM answered specific questions, including:

- What areas in the LDW are net depositional, net erosional, or in dynamic equilibrium?
- How does the composition of the surface layer sediment change over time as external sediment loads (from upstream and lateral sources, such as SDs) become incorporated into the sediment bed?
- What is the effect of high-flow events on episodic scour?
- What is the potential depth of scour during high-flow events?

In addition, the effects of high-flow events on bed scour and the potential for re-exposing buried sediments were evaluated with the STM. The high-flow event simulations were short-term, covering several-day periods equivalent to the storm length and the time necessary for flows to return to normal. For episodic high-flow events, questions of interest included:

- What areas experience erosion during a high-flow event?
- In the areas that experience erosion during high-flow events, what is the potential depth of scour?
- What is the potential for re-exposing buried sediments?

Note that the STM models the movement and fate of suspended and bed load sediment (as physical particles) but does not address chemical transport and fate.

3.2.2 Development of sediment transport model inputs

Input parameters for the STM were separated into three broad categories: 1) sediment characteristics, 2) bed characteristics, and 3) boundary conditions. Sediment characteristics include physical parameters, such as effective particle diameter and settling speed. Bed characteristics include physical parameters of the bulk sediment bed, such as dry density and sediment particle size, and calculated parameters such as erosion rates. Boundary conditions refer to characteristics at the edge of the modeling area (e.g., sediment loads at different inflow locations adjacent to the LDW).

Sediment particles in the LDW have a range of sizes, from less than 1 μm for clays to greater than 2,000 μm for gravel. Simulation of the entire particle size spectrum was impractical. Therefore, particles were separated into four size classes: 1) clay and fine silt (less than 10 µm), 2) medium and coarse silt (10 to 62 µm), 3) fine sand (62 to 250 µm), and 4) medium and coarse sand (250 to 2,000 µm) (Table 3-5). Each sediment size class was represented as an effective particle diameter. The particle size distributions in the sediment bed and incoming loads were evaluated to select the appropriate average effective particle diameters in each general size class to represent particle transport throughout the LDW. Note that effective particle diameters for Classes 1A and 1B were treated as adjustable calibration parameters (Section 3.2.2).

Table 3-5. Characteristics of sediment particle size classes

SEDIMENT SIZE CLASS	PARTICLE SIZE RANGE (µm)	EFFECTIVE PARTICLE DIAMETER (µm)	EFFECTIVE SETTLING SPEED (m/day)
1A. clay, fine silt	< 10	5	1.3
1B. medium, coarse silt	10 – 62	20	21
2. fine sand	62 – 250	130	770
3. medium, coarse sand	250 – 2,000	540	5,500

The settling speeds of sediment particles are related to the effective particle diameter, with settling speed increasing as effective diameter increases. The effective settling speeds for the four sediment size classes had a large range, from about 1 m/day for Class 1A to about 5,500 m/day for Class 3 (Table 3-5). This large range (i.e., factor of approximately 5,000) had a significant effect on the transport characteristics of the different sediment size classes.

The specification of bed properties within the LDW began with the separation of the sediment bed into two distinct types of sediment: cohesive and non-cohesive. Cohesive sediment corresponded to a muddy bed composed of a mixture of clay, silt, sand, and organic matter. Non-cohesive sediment corresponded to a sandy bed with a relatively low amount of clay, silt, and organic matter. The STM required specification of the following bed property inputs within the cohesive and non-cohesive bed areas: 1) dry (bulk) density; 2) initial sediment particle size composition (i.e., relative amounts of Classes 1A, 1B, 2, 3), and 3) effective bed roughness. The median diameter (D_{50}) of particles in the non-cohesive bed areas needed to be specified. The spatial distribution of erosion rate parameters, both horizontally and vertically, for the cohesive bed areas also had to be determined and was derived from an analysis of the Sedflume core data (Section 3.1.5). A summary of model input parameters is presented in Table 3-6, which includes the data sources and an estimate of the level of uncertainty for each input.

Table 3-6. STM input parameters and data sources

MODEL INPUT PARAMETER	Data Source	Level of Uncertainty
Bathymetry and geometry	2003 multi-beam survey of the LDW (Windward and DEA 2004); USACE (bathymetry charts as contained in USACE file number E-12-2.1-12.3, dated September 15, 2000) and Boeing bathymetry data (Pentec et al. 2001)	Measurement uncertainty in vertical bed elevation is: ± 0.5 ft in LDW; ± 2 ft upstream of RM 4.7; ± 1 ft downstream of RM 0.0.
Green River flow rate	USGS gaging station at Auburn (USGS 2007b)	USGS rates discharge data at this station as good (i.e., 10% accuracy).
Tidal elevation in Elliott Bay	NOAA gaging station at Colman Dock (ferry terminal)	Accuracy of tidal elevation measurements is 0.3 cm.
Sediment bed – erosion parameters	Sedflume study conducted during December 2006 (18 cores) (Windward and QEA 2008)	Uncertainty/accuracy of Sedflume core data cannot be assessed. Potential uncertainties related to spatial variability were addressed through a sensitivity analysis.
Sediment bed – dry density	2006 subsurface core data (16 samples) (Windward and RETEC 2007)	The 95% confidence interval, with respect to average value, is 0.14 g/cm^3.
Sediment bed – non-cohesive bed D_{50}	1991 to 2006 core data (875 samples) (see Section 4)	The 95% confidence interval, with respect to average value, is 170 μm.
Sediment bed – effective bed roughness (D_{90})	1991 to 2006 core data (875 samples) (see Section 4)	The 95% confidence interval, with respect to average value, is 110 μm.
Upstream sediment load – magnitude	USGS sediment load studies conducted during 1965-1966 (Harper-Owes 1981) and 1995-1996 (Embrey and Frans 2003)	Annual sediment load estimates have an approximate factor-of-2 level of uncertainty.
Upstream sediment load – composition	USGS sediment load studies conducted during 1965-1966 (Harper-Owes 1981) and 1995-1996 (Embrey and Frans 2003) (composition was adjusted during calibration)	Available data were used to constrain the calibration value to within a realistic range.
Lateral source sediment load – magnitude	CSOs: Approximately 100 TSS samples collected from 1995 to 1997 from five major outfalls (i.e., Brandon, Chelan, Hanford, Connecticut, and King) (King County 1999e).Monthly CSO discharge volumes from June 1999 to May 2006 for nine locations within the LDW were obtained from annual CSO reports and summarized (Nairn 2007). Wet seasons from 1999-2000 and 2005-2006 were included. Storm Drains: Data from over 500 stormwater samples collected at 24 different locations across the US to determine representative TSS concentrations in urban stormwater (SPU 2007). Precipitation data (1986 to 2005) collected at E Marginal Way S pump station were used in watershed model to estimate monthly average volume discharge at each storm drain location (SPU 2007).	The 95% confidence interval, with respect to average value, is 150 mg/L for King County CSOs. Raw data for storm drains were not available, so no confidence interval could be calculated.

MODEL INPUT PARAMETER	Data Source	Level of Uncertainty
Lateral source sediment load – composition	CSOs: GSD estimated from settling analysis results at four CSOs: Denny, Henderson, M.L. King, and Norfolk (Nairn 2007). Storm Drains: GSD data collected from stormwater samples at 16 sites across the US (SPU 2007)	Available data were used to constrain the calibration value to a realistic range.

CSO – combined sewer overflow

GSD – grain size distribution

LDW – Lower Duwamish Waterway

NOAA – National Oceanic and Atmospheric Administration

RM – river mile

TSS – total suspended solids

USACE – US Army Corps of Engineers

USGS – US Geological Survey

Sediment loads from upstream (i.e., Green River) and lateral sources (i.e., SDs, CSOs, streams), including both the magnitude and composition of the loads (i.e., relative amounts of Classes 1A, 1B, 2, 3), were needed to establish boundary conditions for the STM. Loads from the lateral sources were not included during the calibration period simulation (see Section 3.2.3) because the annual average load from lateral sources totaled 0.6% of the upstream load (QEA 2008).

3.2.3 Model calibration and validation

The hydrodynamic model[32] that was originally refined and calibrated during the sediment transport analysis (Windward and QEA 2008), as described in Section 3.1.6, was re-calibrated because modifications were made to the numerical grid.[33] These modifications were necessary so that long-term, multi-year simulations could be completed in a timely manner. In addition, grid cells were added for the LDW slips.

The re-calibration results showed that the hydrodynamic model realistically simulated all of the major characteristics of estuarine circulation in the LDW. The vertical structure of tidal current velocity was realistically simulated, and the STM was able to reproduce the two-layer flow in the region occupied by the saltwater wedge. Strong vertical stratification of salinity has been documented and the model was able to simulate that stratification with acceptable accuracy. In addition, the model realistically simulated the dynamic nature of the saltwater wedge and the location of the toe of the wedge as it varies over the course of a tidal cycle, as well as variations that result from changes in freshwater inflow and the phase of the tidal cycle (i.e., spring and neap tide conditions). Overall, re-calibration of the hydrodynamic model

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[32] This hydrodynamic model (EFDC) was an important component of the STM. The STM differs from EFDC in that it includes sophisticated algorithms for tracking sediment particles, whereas the hydrodynamic portion of EFDC includes only water movement.

[33] As summarized in Section 3.1.6, additional model cells were added to the hydrodynamic model originally constructed by King County to improve the resolution. Because model results are created independently for each grid cell, smaller grid cells allow the evaluation of smaller scale variability in model results.

demonstrated that the model is sufficiently accurate and reliable for the objectives of the STM study.

The STM was built on a foundation of mechanistic formulations that are used to simulate the erosion and deposition of cohesive and non-cohesive sediment. The erosion and deposition formulations used in the model were based on results from a large number of laboratory and field studies. Site-specific data were used to determine model input parameters, with robust datasets used to specify the two major inputs: sediment loading from the Green River and erosion properties of LDW sediments. The significant amount of site-specific data, in conjunction with the mechanistic nature of the model formulations, provided an opportunity to develop and calibrate a reliable STM.

Calibration and validation of the STM was a process that involved comparing model estimates to LDW data. These comparisons provided a "reality check" for the STM and were used to evaluate the accuracy and reliability of the model. The first step in the process was calibration of the STM, which involved adjusting model input parameters to optimize the agreement between model results and empirical data. The second step in the process was model validation, which involved a comparison of model predictions to an independent dataset (i.e., data not used to calibrate the model), with no adjustment of calibration input parameters. Taken together, the results of the model calibration and validation results provided an objective method for "ground-truthing" the STM and determining its reliability as a diagnostic tool to evaluate sediment transport processes in the LDW.

The STM was calibrated and validated over a 21-year period (1960 through 1980) to evaluate the accuracy and reliability of a model. This time period tested the model's capabilities because of the wide range of tidal and river flow conditions during that period, including several high flow events. This time period was selected because the data were readily available in a format that was compatible with the STM. Flows in the Green River during the 1960-to-1980 time period were very similar to flows during 1981-to-2006 time period, based on a comparison of flow duration curves for the two periods. Thus, using the 1960-to-1980 period for model calibration provided results that were consistent with current conditions. For the STM, the calibration targets (i.e., the parameters used to assess model accuracy or agreement) were net sedimentation rates in the navigation channel and bench areas of the LDW (QEA 2008). Four model parameters were adjusted to achieve the optimum agreement between estimated and empirically derived estimates of net sedimentation rates in the navigation channel and bench areas (Table 3-7) (QEA 2008).

Table 3-7. STM calibration parameter values

Adjusted Parameter	Calibration Value
Effective particle diameter of Class 1A sediment	5 µm
Effective particle diameter of Class 1B sediment	20 µm
Average percentage of Class 1A sediment in incoming suspended load[a]	70%
Average percentage of Class 1B sediment in incoming suspended load[a]	18%

[a] The other 12% of the incoming suspended load consisted of Class 2 and 3 sediments.

STM – sediment transport model

In addition to the parameters listed in Table 3-7, two additional adjustments were made to the STM during calibration. First, a particle-shielding factor was incorporated into the calculation of bed erosion flux. This factor was used to reduce the erosion flux of smaller particles that can be sheltered by larger particles within a graded bed (i.e., a bed with a wide range of particle sizes) (QEA 2008). The second adjustment to the model was to specify that the sediment bed upstream of RM 4.8 be treated as a "hard bottom," which means that the bed in that region (i.e., in the river upstream of the Upper Turning Basin) was assumed to not experience erosion or deposition. Bed load transport of sediment was allowed in that region (QEA 2008).

Validation of the model was achieved through comparison of the estimated composition of sediment deposited in the LDW to the observed composition of surface-layer sediment. The relative amounts of the four size classes (i.e., Classes 1A, 1B, 2, 3) in the sediment estimated to be deposited between RM 0.0 and RM 4.3 (i.e., region downstream of the Upper Turning Basin) during the 21-yr calibration period were compared with the relative amounts of the four sediment size classes measured in the Sedflume cores, which represent the upper 30 cm of the sediment bed. No model input parameters were adjusted to optimize the agreement between observed and estimated composition. The satisfactory model results provided additional support for the capability of the model to realistically and accurately simulate erosion and deposition processes in the LDW.

Results of the calibration and validation exercises indicated that the STM is able to realistically and accurately simulate erosion and deposition processes in the navigation channel and bench areas of the LDW. The calibration and validation results, coupled with constraints on the model by mechanistic formulations and site-specific data, indicated that the model is a reliable tool for evaluating sediment transport processes in the LDW over a wide range of spatial and temporal scales. Quantitative uncertainty estimates have been made for STM predictions, providing additional data to assess the degree to which the model output is considered valid at various scales. Over small spatial scales, the STM can typically be used as one line of evidence, along with other information and data, to guide decision-making. In addition, the STM provides a reliable framework to extrapolate conditions for areas without erosion and/or empirical net sedimentation rate data.

The calibration and validation of the STM resulted in a revised definition of reach boundaries from those presented in the STAR. The reach designations discussed in this section, and in the STM, represent a refinement, based on an improved understanding of LDW sediment transport processes, from the modeling conducted for the STAR. The revised reach definitions were:

- **Reach 1.** Located downstream of RM 2.2 and includes the saltwater wedge during all flow and tidal conditions.
- **Reach 2.** Extends from RM 2.2 to RM 4.0 and includes the saltwater wedge toe.
- **Reach 3.** Located from RM 4.0 to RM 4.8, with flow in the uppermost portions of this reach corresponding to a freshwater tidal river under most flow conditions. This reach is occupied by the saltwater wedge only during low- and average-flow conditions; under high-flow conditions, the entire reach may behave as a freshwater tidal river.

3.2.4 Sediment bed stability results

The STM was used to evaluate the effects of high-flow events on bed stability in the LDW (QEA 2008). High-flow events with return periods of 2, 10, and 100 years were simulated. Boundary conditions for the high-flow simulations were developed using the hydrograph of a high-flow event that occurred during November-December 1975. During this high flow event, spring tide conditions occurred during peak flow conditions. Thus, the high-flow simulations produced conservative results that represented upper-bound estimates of bed scour depths during high-flow events.

Two quantities related to bed scour were derived from the simulation results: net erosion and maximum bed scour depth (QEA 2008). Net erosion at a specific location is the total decrease in bed elevation over the course of a high-flow event (i.e., difference in bed elevation between start and end of 26-day simulation). Net erosion thus incorporates the effects of erosion and deposition during the entire event. Maximum bed scour depth is the maximum depth of erosion that occurs at a specific location during the high-flow event. Typically, maximum bed scour occurs near the time of the peak flow rate, with deposition occurring after the peak flow rate, as river discharge declines.

Spatial distributions of estimated net erosion at the end of high-flow events with return periods of 2, 10, and 100 yrs are presented on Maps 3-4, 3-5, and 3-6, respectively. The spatial distribution of maximum bed scour during the 100-year high-flow event is shown on Map 3-7. The locations of the maximum net erosion depths of bed scour in cohesive and non-cohesive bed areas are denoted on these maps. Net erosion, where it was predicted to occur (i.e., above RM 2.0), was generally estimated to be higher in the navigation channel than on the benches.

Net erosion was estimated to occur over approximately 15 to 20% of the area of the LDW sediment bed during a high-flow event. Net erosion of 10 cm or greater was estimated to occur over about 6% (22 ac) of the bed area in the LDW during the 100-year high-flow event. Maximum net erosion depths during a 100-year event were estimated to be about 21 cm. Maximum bed scour depths were about 1 cm greater than net erosion depths. Thus, based on the model, scour was estimated to be limited to approximately the upper 20-cm layer of the sediment bed within a few small areas during the 100-year high-flow event upper-bound estimate. In most areas, the maximum scour depth was estimated to be far less. Even during a 100-year high-flow event, negligible bed scour was estimated to occur over large areas of the LDW, including nearly all of Reach 1 (Map 3-7).

The model results showed that net erosion primarily occurred between RM 2.2 and RM 4.0 during the high-flow event simulations in the LDW, including the 100-year event (QEA 2008). Most of the net erosion occurred in this reach of the LDW because most of the region upstream of approximately RM 2.2 behaved as a freshwater tidal river during high-flow events, with the saltwater wedge toe located downstream of approximately RM 2.2 during ebb tides. This is consistent with the hydrodynamic model simulation, which showed that relatively high bed shear stresses occurred upstream of the saltwater wedge (Windward and QEA 2008). Less net erosion was estimated to occur in the reach upstream of RM 4.0 than in the reach from RM 2.2 to RM 4.0 for two reasons. First, bed shear stresses were generally lower upstream of RM 4.0 than in the reach from RM 2.2 to RM 4.0, which is mainly the result of differences in cross-sectional area between the two reaches (i.e., larger cross-sectional area upstream of RM 4.0). Second, more deposition and less net erosion were estimated to occur upstream of RM 4.0 because a large of portion of sand (i.e., sediment Classes 2 and 3) transported from the Green River to the LDW was estimated to be deposited in this reach; the Upper Turning Basin was designed to be an effective sediment trap.

Net erosion was estimated to occur in only a few relatively small areas in the reach downstream of RM 2.2, even during a 100-year event, with typical net erosion depths of 2 cm or less (Map 3-6). The cause for the significant difference in estimated erosion depths between the reaches upstream and downstream of RM 2.2 is the presence of the saltwater wedge downstream of RM 2.2 during high-flow events, which resulted in relatively low bed shear stresses and a small amount of erosion in that reach.

The effects of deposition on net erosion after the peak flow rate, as river discharge declines, are illustrated by a comparison of the spatial distributions shown on Maps 3-6 and 3-7. These two maps show that: 1) the areal extent of maximum bed scour was estimated to be larger than the areal extent of net erosion at the end of the 100-year event, 2) net erosion was estimated to be less than maximum bed scour at most locations, and 3) net deposition over the course of the event was estimated to occur at some locations that experienced bed scour at some time during the event. In areas that experienced deposition after the maximum bed scour occurred (i.e., after the peak flow during the event), the amount of deposition was typically relatively small.

The relative amounts of sediment eroded from surface (0 to 10 cm) and subsurface (deeper than 10 cm) layers during the 100-year event were evaluated by "tagging" sediments within those two layers at the start of the simulation and tracking the sediment from those two bed sources separately (QEA 2008). Results of a mass balance analysis for the 100-year event indicated that 20% of the mass of sediment eroded from the sediment bed in the LDW was from the subsurface layer (i.e., deeper than 10 cm), with the remaining 80% of eroded sediment originating from the surface layer (0 to 10 cm). The mass balance analysis also showed that about 4 and 2% of the total sediment transported downstream of RM 0.0 during the 100-year event was eroded from the surface and subsurface layers, respectively (the remainder coming from the Green River). These results indicated that subsurface contamination (i.e., >10 cm depth) is unlikely to be significantly eroded and transported within and outside the LDW during a high-flow event.

A sensitivity analysis was conducted to evaluate uncertainty in model predictions that may have resulted from uncertainty in model input parameters. The model was most sensitive to the parameters that control erosion rates, which were specified using Sedflume core data collected from the LDW. Based on model input uncertainty, uncertainty in the extent of areas estimated to have erosion was less than ± 50% within RM 0.0 to RM 4.3, relative to the base-case simulation. Uncertainty in predicted sediment mass eroded ranged from about -50 to +75% within RM 0.0 to RM 4.3 as well as in the east bench and navigation channel. The predicted sediment mass eroded in the west bench zone is more uncertain; the uncertainty range in the west bench was about -40 to +130%.

3.2.5 External sediment load analysis, net sedimentation rate analysis, and changes in bed composition

A 30-year simulation was conducted so that the effects of external sediment loads on surface layer bed composition over long-term, multi-year periods could be evaluated (QEA 2008). In addition, net sedimentation rates that account for all sources of sediment were estimated. Boundary conditions for the model were specified using river flow rate and tidal elevation data collected during the 30-year period from 1960 through 1979. External sediment loads were specified from two sources: 1) upstream loads (i.e., Green River), and 2) lateral loads (i.e., SDs, CSOs, and streams). Original bed sediment (i.e., sediment at the beginning of the 30-year simulation) was treated as a third source of sediment. The effects of the external loads on the relative amounts of sediment from the three sources (i.e., bed, upstream, and lateral) in the surface layer of the sediment bed over the 30-year period were estimated by the STM. The surface layer was defined as the top 10 cm of the bed.

Sediment from each of the three sources was separated into four particle size classes (i.e., Classes 1A, 1B, 2, 3), with the sediment transport characteristics of the four particle size classes being the same for all three sediment sources (QEA 2008). For example, the erosion, deposition, and transport of Class 1A sediment was treated the same way for sediment originating from the bed, upstream, and lateral sources. Thus, the model simulated the erosion, deposition, and transport of 12 sediment classes (i.e., four particle size classes from each of the three sediment sources) during the 30-year period.

Changes in the composition of the 10-cm layer at the surface of the sediment bed were of particular importance in this analysis. At beginning of the 30-year simulation, the composition of the 10-cm surface layer was 100% bed-source sediment, with no sediment from upstream and lateral sources. As the 30-year simulation progressed, upstream- and lateral-source sediments were transported in the LDW and deposited on the 10-cm surface layer, which reduced the relative amount of bed-source sediment in that layer. The model tracked spatial and temporal changes in the relative amounts of sediment from the three sources over the course of the 30-year period that resulted from erosion, deposition, and transport processes in the LDW.

3.2.5.1 Sediment load and mass balance

Sediment loads from lateral sources (i.e., SDs, CSOs, and streams) were estimated using an approach that combined watershed modeling and empirically derived estimates (QEA 2008; SPU 2007; Nairn 2007). SDs, streams, and CSOs discharge sediment into the LDW at over 200 locations; incorporating each individual discharge location into the model was not practical. Thus, the lateral sources were aggregated and represented by 21 point sources that discharged into the LDW at 16 representative locations (Map 3-8). The total average annual load from lateral sources was estimated to be approximately 1,200 metric tons per year (MT/yr), with 76%, 3%, and 21% of the total lateral load from SDs, CSOs, and streams, respectively (QEA 2008). The total sediment load from lateral sources over a 30-year period (32,600 MT) was 0.5% of the total load from the upstream (river) source (6,266,400 MT) [34] (Figure 3-2).

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[34] Note that the values on Figure 3-2 are totals for the 30-yr modeling period, not annual averages.

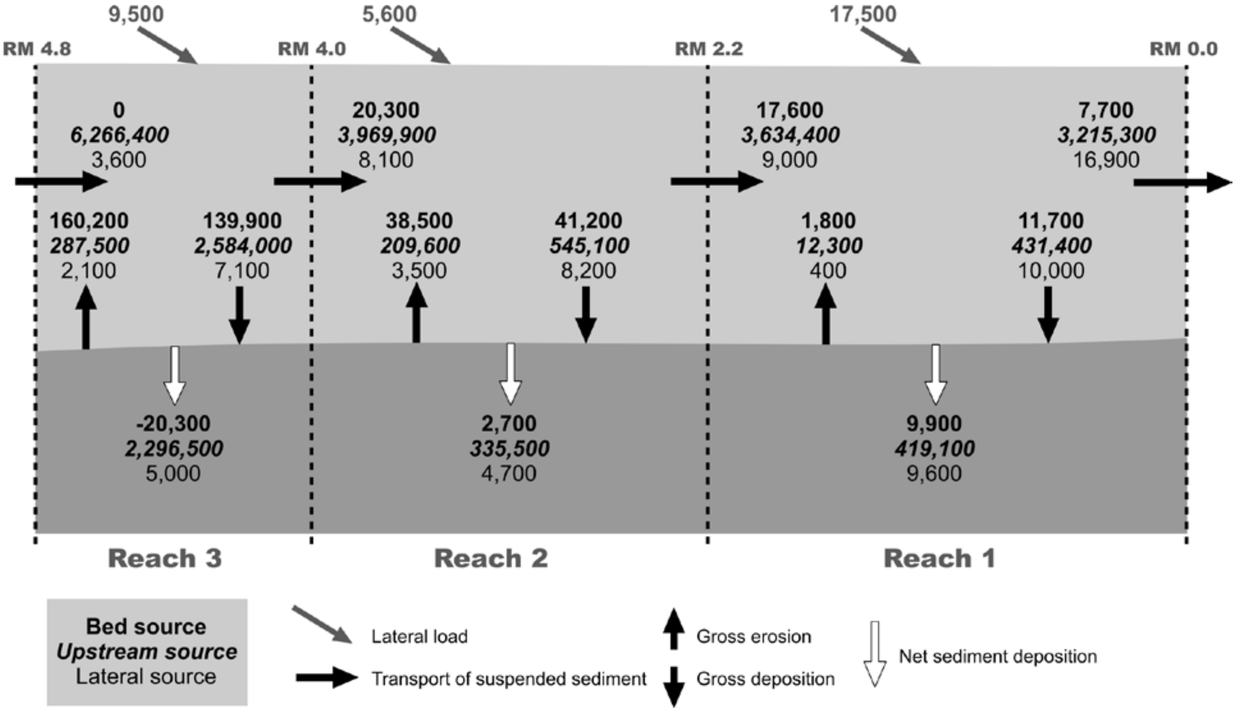

Figure 3-2. Sediment mass balances (in metric tons) for bed-, upstream-, and lateral-source sediment in the LDW (RM 0.0 to RM 4.8) for a 30-year period

3.2.5.2 Net sedimentation rates

The spatial distribution of average net sedimentation rates estimated for the 30-year period is presented on Map 3-9. These model results illustrate several characteristics of sediment transport processes in the LDW (QEA 2008). Reach 1 (RM 0.0 to RM 2.2) was estimated to be net depositional over multi-year periods with net sedimentation occurring everywhere. Within Reach 1, relatively high net sedimentation rates (i.e., greater than 2 cm/yr) were estimated between RM 1.4 and RM 2.2, with lower rates (i.e., typically 0.5 to 2 cm/yr) estimated downstream of RM 1.4. The model estimated a relatively low net sedimentation rate (i.e., less than 0.1 cm/yr) in a small area near RM 0.8 to RM 0.9 over the 30-year period, with net erosion during high-flow events (Section 3.2.4). Within Reach 2 (RM 2.2 to RM 4.0), net sedimentation rates were estimated to be generally lower than those in Reaches 1 and 3 (RM 4.0 to RM 4.8). Net erosion was predicted for three Reach 2 grid cells in the STM. This small area of net erosion over the 30-year period appeared to be in a state of approximate dynamic equilibrium. Relatively high variability in net sedimentation rates, from less than 0.5 cm/year to greater than 3 cm/yr, in Reach 2 reflected the dynamic nature of this reach, which was estimated to experience episodic bed scour during high-flow events (Section 3.2.4). Reach 3 was estimated to be net depositional and had the highest net sedimentation rates within the LDW, which is not surprising because the Upper Turning Basin, which is designed to function as a sediment trap, is located in this reach. The maximum net sedimentation rate estimated by the model (approximately 140 cm/yr) was for several grid cells in the Upper Turning Basin.

Large-scale comparisons of net sedimentation rates among the three reaches were made by calculating reach-averaged values (Table 3-8). Reach 2 was separated into two sub-reaches to reflect significant spatial variations in net sedimentation rates within that reach (QEA 2008). Sedimentation rate averages for each reach reflect model results for both benches and the navigation channel. In Reaches 1 and 2, the average net sedimentation rates estimated for the benches and navigation channel were similar (1.8 to 3.7 cm/yr). For Reach 3, the 30-year net sedimentation rate for the navigation channel was more than double the rate for either the west or east bench. The value for Reach 3 corresponds to the average in the area excluding the Upper Turning Basin; average net sedimentation in the Upper Turning Basin was over 40 cm/yr.

Table 3-8. Reach-average sedimentation rates and composition at end of 30-year period

REACH	NET SEDIMENTATION RATE (cm/yr)	BED-SOURCE CONTENT (%)	UPSTREAM-SOURCE CONTENT (%)	LATERAL-SOURCE CONTENT (%)
1 (RM 0.0 to RM 2.2)	1.8	3	94	3
2 (RM 2.2 to RM 4.0)	2.8	13	85	2
2A (RM 2.2 to RM 2.6)	3.7	3	96	1
2B (RM 2.6 to RM 4.0)	2.4	17	81	2
3 (RM 4.0 to RM 4.8, excluding the Upper Turning Basin)	15	3	95	2

RM – river mile

For the navigation channel, the estimated net sedimentation rate decreased from the Upper Turning Basin (near RM 4.5) to downstream areas, primarily because less sand is transported to the downstream reaches than to the upstream reach. The Upper Turning Basin acts as a sink for sand from upstream sources, but silt and clay remain in suspension longer and generally settle farther downstream compared with sand. A similar phenomenon occurs for lateral inflows to the LDW. Based on model results, approximately one-quarter of the silt and almost all the clay is discharged to Elliott Bay, regardless of the source.

Net sedimentation rates can be used to estimate the time necessary to add to the existing surface sediment layer to a specified thickness. For example, the time needed to add 10 cm of new sediment to the existing surface sediment layer (which is defined as 0 to 10 cm) is estimated as:

$$T = 10/\text{net sedimentation rate} \qquad \textbf{Equation 3-5}$$

where T is time measured in years, and the net sedimentation rate is measured in cm/yr. This simplified approach does not account for the physical mixing of sediment from different sources during deposition and erosion processes, nor for consolidation with time.

In general, net sedimentation rates estimated using the empirical approach (see Section 3.1.4) agreed reasonably well with the model estimates, considering the different methods used to derive the estimates. Of the 63 locations where a comparison between empirical (Map 3-3) and estimated (Map 3-9) net sedimentation rates was possible, 40 locations had a difference of 1.0 cm/year or less (Table 3-9). The model tended to over-estimate net sedimentation rates in the bench regions upstream of RM 4.0. Between RM 2.0 and RM 4.0, model results were generally within the range of the empirically derived estimates in the east bench area, with a tendency to be above empirical estimates in the vicinity of RM 2.5 to RM 2.8. Within the west bench area, between RM 2.0 and RM 4.0, the model tended to over-predict empirically derived estimates of net sedimentation rates. Downstream of RM 2.0 in both bench areas, the model results were less variable than the empirically derived estimates, but the model results were in agreement with the general trend of the empirically derived estimates.

Table 3-9. Comparison of net sedimentation rates derived using empirical data and the STM

CORE ID[a]	RIVER MILE	NET SEDIMENTATION RATE (cm/yr)		WERE RATES WITHIN 1 cm/yr OF EACH OTHER?
		EMPIRICAL ESTIMATE[b]	MODEL ESTIMATE[b]	
SC1	0.0	0.9 – 1.7	1.3	yes
SC3	0.1	0.4	1.2	yes
SC5	0.1	0.5 – 0.9	1.3	yes
Sg1a	0.2	0.9 – 1.3	1.2	yes
SC4	0.2	1.1 – 1.7	0.76	yes
B3	0.2	4.9 – 5.1	1.3	no
SC6	0.3	2.3 – 3.0	1.1	no
SC8	0.4	1.2 – 3.3	1.1	yes
SC7	0.4	0.7	1.1	yes
DUD006	0.5	1.9 – 3.1	4.0	yes
SC9	0.5	1.8	1.2	yes
SC10	0.5	2.4 – 2.9	1.5	yes
SC11	0.5	0.4 – 0.5	1.3	yes
SC12	0.6	1.2 – 2.9	1.7	yes
Sg2	0.7	0.5 – 0.6	1.1	yes
SC13	0.9	1.1 – 2.1	1.6	yes
SC14	0.9	2.9 – 4.0	0.0	no
SC15	0.9	1.4 – 4.8	1.7	yes
SC16	1.0	1.2 – 3.8	1.9	yes
SC17	1.0	2.9	1.9	yes
SC18	1.0	0.7 – 1.9	2.1	yes

Core ID[a]	River Mile	Net Sedimentation Rate (cm/yr)		Were Rates Within 1 cm/yr of Each Other?
		Empirical Estimate[b]	Model Estimate[b]	
SC19	1.0	3.0 – 4.7	1.5	no
SC21	1.0	2.3 – 4.9	1.5	yes
Sg3	1.2	1.9 – 2.1	2.3	yes
SC23	1.2	3.3 – 4.8	2.5	yes
SC24	1.2	0.7 – 1.1	1.8	yes
SC25	1.3	1.2 – 3.0	2.0	yes
Sg4	1.4	1.6 – 2.0	2.5	yes
SC27	1.4	0.9 – 2.6	2.6	yes
SC29	1.4	0.4 – 0.6	2.5	no
SC30	1.6	1.1	2.5	no
SC31	1.7	1.0 – 1.5	2.5	yes
SC32	1.7	1.7 – 2.5	2.5	yes
S6	1.8	2.2 – 3.2	4.7	no
SC34	1.9	2.2	0.26	no
SC33	1.9	2.9 – 3.8	2.6	yes
Sg5a	1.9	1.4 – 1.6	0.45	yes
SC35	2.0	2.8 – 3.7	3.1	yes
SC36	2.1	2.2 – 2.8	2.3	yes
SC37	2.2	1.0 – 2.6	1.8	yes
SC39	2.2	2.9	3.3	yes
S13	2.2	1.5	3.4	no
SC40	2.3	0.7	2.8	no
Sg6	2.3	2.5 – 2.7	1.5	yes
SC41	2.4	2.6	2.0	yes
SC42	2.5	2.7	4.7	no
SC43	2.6	0.5 – 3.0	5.2	no
SC44	2.7	0.3 – 1.4	0.21	yes
Sg7	2.7	1.9 – 2.1	0.17	no
SC46	2.7	1.8 – 2.3	5.6	no
SC-11	2.9	1.5 – 2.2	2.5	yes
SC47	3.0	1.0 – 2.2	0.02	yes
SC48	3.3	0.4 – 0.5	2.1	no
SC49	3.5	2.4 – 4.3	2.3	yes
Sg9	3.6	0.3 – 0.9	6.9	no
Sg10	3.6	1.6 – 1.8	3.7	no
SC50	3.8	0.9 – 1.5	0.06	yes
S3	3.8	3.0 – 3.3	3.5	yes
SC51	3.8	0.6	2.3	no
Sg11b	3.9	0.6 – 0.7	1.7	yes
SC52	3.9	0.5 – 0.9	1.7	yes

CORE ID[a]	RIVER MILE	NET SEDIMENTATION RATE (cm/yr)		WERE RATES WITHIN 1 cm/yr OF EACH OTHER?
		EMPIRICAL ESTIMATE[b]	MODEL ESTIMATE[b]	
SC53	4.2	3.1 – 3.3	2.2	yes
SC54	4.3	1.8 – 2.7	11	no
Sg12	4.3	> 2.0	11	no
Sg13	4.4	2.3 – 2.6	61	no
SC56	4.7	0.8 – 1.0	24	no
SC55	4.9	0.3 – 1.0	34	no

[a] Core IDs are shown on Map 3-3.

[b] Net sedimentation rates (both empirical and modeled) are shown on Map 3-3. This map shows the modeled rates as ranges, but single values were obtained from the STM output for use in this table.

ID – identification

STM – sediment transport model

3.2.5.3 Bed composition

A primary objective of this analysis was to evaluate the effects of external sediment loads on the composition of the surface layer (0 to 10 cm) sediment. This type of analysis will be used in the FS to estimate the rate and magnitude of sediment quality improvements in the future. Large-scale comparisons of the three reaches provided an assessment of the relative effects of upstream and lateral sediment loads on surface layer composition over the course of the 30-year period (Table 3-8). As shown in Table 3-8, the sediment bed will be composed mostly of upstream solids from the Green River (81 to 96% composition) over time, with less than 2% of the load coming from lateral sources.

The spatial distribution of bed-source content in the surface layer at the end of the 30-year simulation period is shown on Map 3-10. The relatively high net sedimentation rates estimated in Reaches 1 and 3 resulted in significant estimates of decreases in bed-source content (QEA 2008). The spatial distribution of upstream-source content in surface layer sediment at the end of the 30-year period was a mirror image of the bed-source content; upstream-source content was high at locations where bed-source content was low and vice versa (Map 3-11). In general, the estimated effects of sediment loads from lateral sources were greatest in the immediate vicinity of the discharge points of creeks, SDs or CSOs (Map 3-12). However, it is important to note that over 200 SDs and CSOs were represented by only 21 point sources that discharged into the LDW at 16 discrete locations in the model. Therefore, the true lateral-source content in the surficial bed layer is expected to be more widely distributed.

In Reaches 2 and 3, lateral-source contributions were greater than 1% in surface layer sediment in the slips near the SD/CSO discharge points. This effect was seen in Slips 4 and 6, located at approximately RM 2.9 and RM 4.2, as well as in the bench areas that extend downstream of near-shore discharge locations. The effects of lateral sources on surface layer composition were more widely distributed in Reach 1, with lateral-source contributions greater than 1% in a large portion of this reach. Tidal effects within this reach were estimated to spread lateral-source sediment upstream and downstream of the discharge locations. Relatively high lateral-source content values (i.e., greater than 5%) were estimated in the grid cells where the lateral-load discharges were specified. These contributions are likely lower than the estimates because of the aggregation of inputs.

The spatial distributions of surface layer composition shown on Maps 3-10 through 3-12 represent conditions at the end of the 30-year simulation. Temporal variations in surface layer composition over the entire 30-year period indicated that bed-source content is estimated to generally decrease at an approximately exponential rate (QEA 2008).

Sediment mass balances were constructed for the 30-year period to gain additional insights about the movement within the LDW of sediment from bed, upstream, and lateral sources. The details of the mass balance calculations were presented in QEA (2008) and are summarized briefly here. The overall trapping efficiency (TE) for the LDW (RM 0.0 to RM 4.8) was 49% (Figure 3-3). TE is the portion of the incoming sediment load that is deposited within a particular region. TE varied among the three reaches because of differences in the hydrodynamic and sediment transport characteristics of those reaches. The TE of Reach 3 was 36%, which was higher than TE values of 12% and 9% for Reaches 1 and 2, respectively (Figure 3-3). The high TE value for Reach 3 was attributable to the presence of the Upper Turning Basin, which is designed to be an efficient sediment trap to capture a large portion of the sand that is transported from the river into Reach 3. The lowest TE value was in Reach 2 because of the hydrodynamic characteristics of this reach during high-flow events. Reach 1 had a higher TE (12%) than did Reach 2 because of minimal erosion during high-flow events. Reach 1 had a substantially lower TE than did Reach 3 because of the relatively small amount of sand transported into Reach 1 compared with the amount of sand transported into and deposited within Reach 3. Overall, about 50% of the sediment load coming into the LDW from upstream (6,270,000 metric tons over 30 years) settles in the LDW; and the remaining load, composed mostly of finer silt-clay particles, stays suspended in the water column and is transported out of the LDW (3,223,900 metric tons) (Figure 3-3). Approximately 10% of the fine silt-clay particles (Class 1a) that enter the LDW system, either from upstream or lateral sources, are deposited in the LDW. In contrast, 76%, 96%, and >99% of the other particle size classes (medium-coarse silt, fine sand, and medium-coarse sand, respectively) from upstream or lateral sources are deposited in the LDW. Figure 3-3 shows the LDW solids mass balance, illustrating the net depositional environment of the LDW.

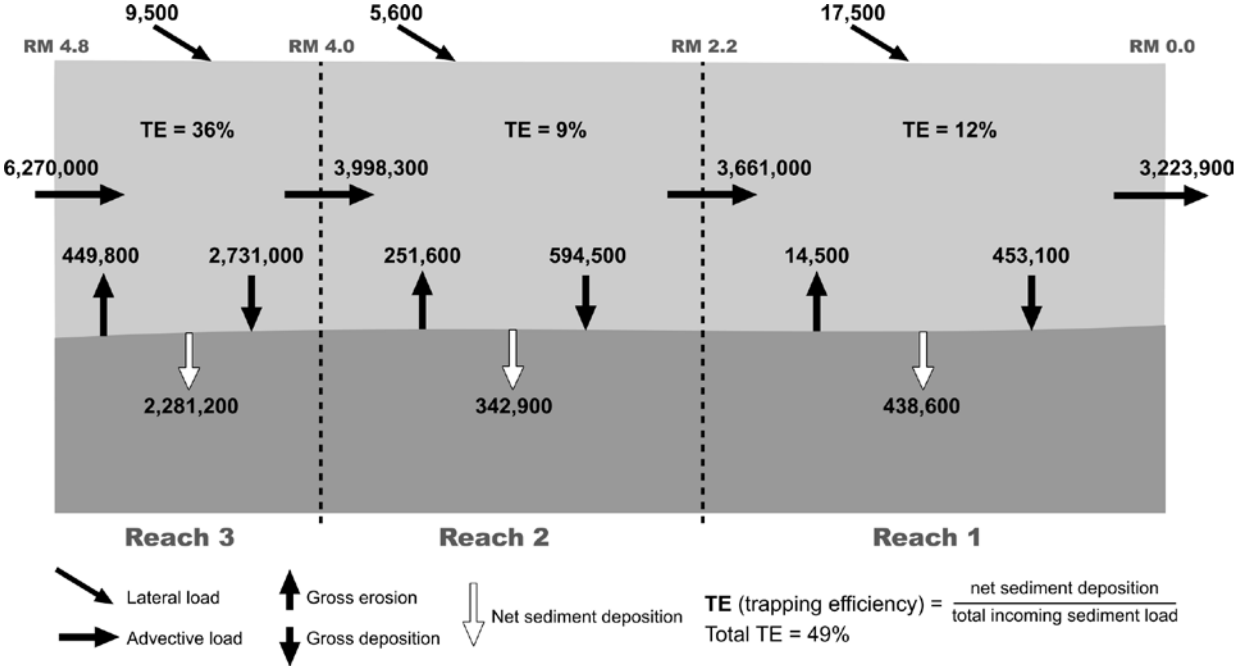

Figure 3-3. Sediment mass balance (in metric tons) in the LDW (RM 0.0 to RM 4.8) for a 30-yr period

A sensitivity analysis was used to evaluate uncertainty in model predictions. The sensitivity of the model to changes in the magnitude and composition of external sediment loads was investigated (see Section 4.4 of the STM report, QEA 2008). The results of the analysis showed that uncertainty in the lateral-source content was about ±40% with respect to the base-case value. The bed-source content varied non-linearly with respect to changes in the upstream-source load magnitude (e.g., doubling the upstream load caused the bed-source content to decrease by about 50%; decreasing the upstream load by 50% caused the bed-source content to increase by about 50%). Changes in the particle size composition of the upstream load resulted in variations that were within 10% of the base-case value. Thus, uncertainty in specifying the magnitude and composition of the upstream- and lateral-source loads produces uncertainty levels of about + 40% and + 50% in the predicted lateral-source and bed-source contents, respectively, in the mixing-zone layer.

3.2.6 Uncertainty analysis

The objective of the uncertainty analysis was to evaluate the effect of uncertainty on model predictions by varying key model input parameters that appeared sensitive to change. Based on the sensitivity analysis results, five model input parameters were selected to quantitatively evaluate model prediction uncertainty: 1) upstream sediment load, 2) settling speeds of Class 1A/1B sediment particles, 3) erosion rate parameters, 4) effective bed roughness, and 5) Class 2/3 particle diameter. Lower- and upper-bound limits for each of these five input parameters were determined, and a factorial analysis was conducted, which resulted in 32 simulations to account for all of the possible combinations of the bounding limits of the five input parameters. Details of the technical approach for conducting the uncertainty analysis are provided in Appendix D of the STM report (QEA 2008).

The uncertainty analysis demonstrated that upstream sediment load and the settling speed of Class 1A/1B sediment particles were the primary controlling factors of predicted net sedimentation rates over multi-year periods in the LDW. The other three input parameters (i.e., erosion rate parameters, effective bed roughness, and Class 2/3 particle diameter) had only minor effects on multi-year model predictions. The upstream sediment load was specified using the results of two USGS studies, one for 1965-1966 (Harper-Owes 1981) and another for 1995-1996 (Embrey and Frans 2003), which provided good estimates of the magnitude of the Green River load. Class 1A/1B settling speeds were treated as adjustable parameters during model calibration; the model was relatively sensitive to these parameters. The settling speed values were estimated as a result of the calibration process and consequently should be relatively accurate. Therefore, the two primary input parameters controlling predicted net sedimentation rates over multi-year periods were reliably defined by site-specific data and model calibration. Additional results from the STM uncertainty analysis are provided in Appendix D of the STM report (QEA 2008).

The results of the uncertainty analysis summarized here were used to generate realistic lower- and upper-bound uncertainty limits on the model calibration results. Comparisons of the realistic bounding limits with the original calibration results for the navigation channel, east bench, and west bench were made in the STM report (QEA 2008). The uncertainty analysis demonstrated that uncertainty in model input parameters does not change the overall STM conclusions or CSM.

3.3 INTEGRATION AND SYNTHESIS OF SEDIMENT TRANSPORT ANALYSIS RESULTS

The analyses conducted for the STM, in conjunction with empirical and ship-induced bed scour analyses, have resulted in an improved understanding of sediment transport processes in the LDW. A large amount of information on LDW hydrodynamics and sediment transport is presented in the STAR (Windward and QEA 2008) and the STM report (QEA 2008), respectively. Results and findings from the major components of the analyses are integrated and synthesized in this section to produce a clear and concise picture of sediment transport in the LDW. Additional integration of sediment transport analysis results is provided in Section 4.3, where temporal trends in chemical concentrations are discussed, and in Section 10.2.2, where sediment chemistry data are discussed in the context of a chemical CSM.

The first step in understanding sediment transport in the LDW was to understand the hydrodynamics of this saltwater wedge estuary. During low-flow conditions in the Green River, the saltwater wedge extends up to or beyond the upstream portion of Reach 3 (i.e., RM 4.5 to RM 4.8), see Figure 3-4. The saltwater wedge is dominated by two-layer estuarine circulation, with saltier and denser water transported upstream in the lower layer of the water column and fresher water transported downstream in the upper layer. Near-bed velocities and bed shear stresses within the saltwater wedge are tidally driven and relatively low, which results in minimal bed scour within the saltwater wedge during low-flow conditions. During a high-flow event in the Green River, the toe of the saltwater wedge is pushed downstream to the vicinity of the boundary between Reaches 1 and 2 (i.e., RM 2.0 to RM 2.5). Two-layer estuarine circulation exists in Reach 1 during high-flow events (Figure 3-5). High freshwater inflow from the Green/Duwamish River causes the hydrodynamic characteristics of Reaches 2 and 3 to change from two-layer estuarine circulation (low-flow conditions) to a freshwater tidal river. This change in the hydrodynamic characteristics of these two reaches results in significant increases in near-bed current velocities and bed shear stresses in Reaches 2 and 3 during a high-flow event.

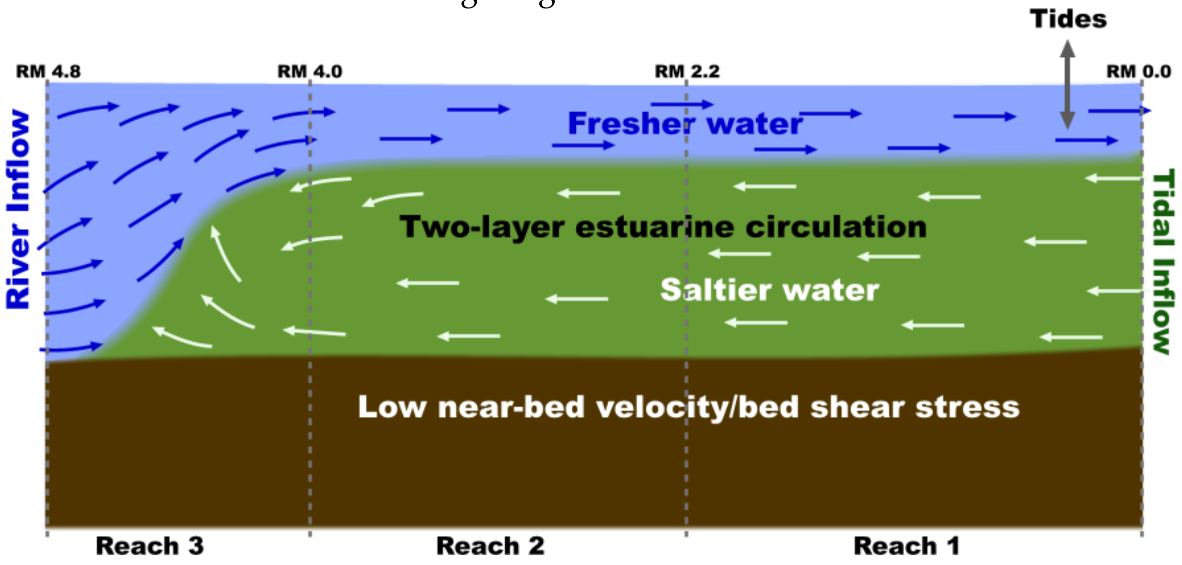

Figure 3-4. Schematic of LDW hydrodynamics during low-flow conditions

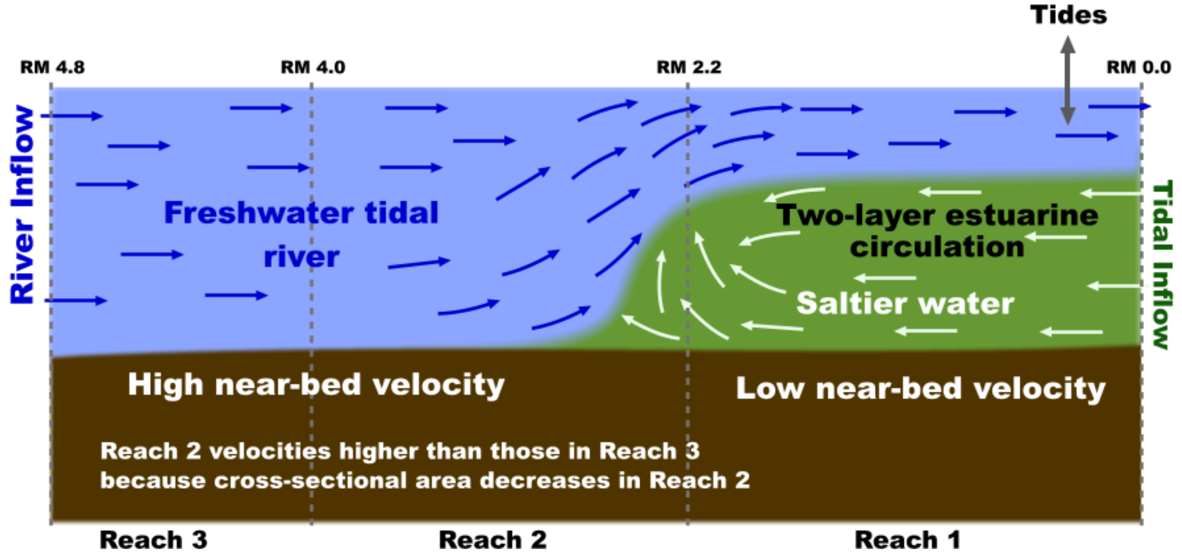

Figure 3-5. Schematic of LDW hydrodynamics during high-flow conditions

3.3.1 Sediment bed stability

Sediment bed stability analyses conducted to date included evaluations of high-flow events, with durations on the order of days to weeks, and long-term simulations over 30 yrs. Overall, model results from both high-flow event and long-term simulations suggested that Reaches 1, 2, and 3 have different sediment transport characteristics, primarily as a result of differences in the hydrodynamic characteristics of each reach. Reach 1 was estimated to have relatively high net sedimentation rate estimates, but these estimates were generally lower than the estimates in Reach 3 because less sand is transported into this portion of the LDW than into the upstream reach. Reach 2 was estimated to have spatially variable net sedimentation rates, with areas of relatively low net sedimentation. This reach experiences the most erosion during a high-flow event as a result of relatively high bed shear stresses caused by changes in hydrodynamics during an event (i.e., transition from estuarine circulation with a saltwater wedge during low-flow conditions to a freshwater tidal river during high-flow conditions). Reach 3 has the highest net sedimentation rate estimates in the LDW, because of the presence of the Upper Turning Basin, which acts as a sediment “sink” and captures a large portion of the load of sand from the Green River.

During a high-flow event, most of the bed scour is estimated to occur in Reach 2, with Reach 1 having minimal erosion, except in a small area near RM 0.8 to RM 0.9. Limited net erosion is estimated in Reach 3. During a high-flow event with a return period of 100 yrs, the model estimated that only about 18% of the total bed area in the LDW (i.e., about 70 ac) was net erosional, with most of the bed scour occurring in Reach 2. The remaining 82% of the total bed area in the LDW was estimated to be net depositional during a 100-year high-flow event (Figure 3-6). A large majority of the net erosion was estimated for the surface layer (i.e., 0 to 10 cm) of the bed, with only about 6% of the total bed area (about 22 ac) having net erosion greater than about 10 cm. The maximum depth of bed scour during a 100-year high-flow event was estimated to be about 21 cm, although this depth of bed scour was estimated for very small areas. Approximately 78% of the sediment mass eroded from the bed during a 100-year high-flow event was estimated to be redeposited within the LDW, with the remaining 22% transported out of the LDW.

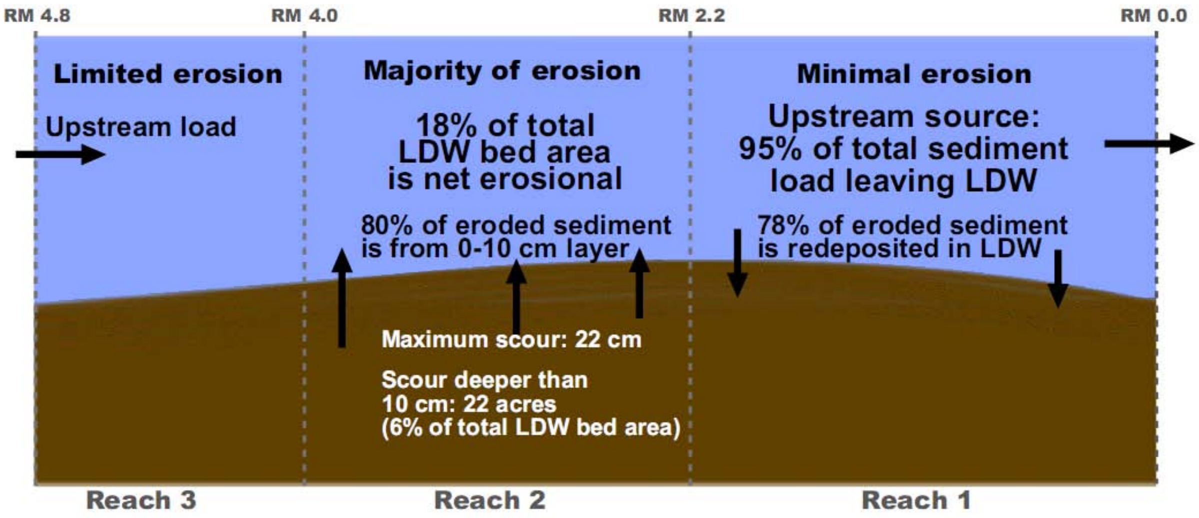

Figure 3-6. Schematic of LDW sediment transport processes during a 100-year high-flow event

Reaches 1, 2, and 3, and thus the entire LDW, were estimated to be net depositional on annual time scales.[35] The general effect of erosion during high-flow events was to reduce the net sedimentation rate in locations where bed scour occurs. Net sedimentation rates on a reach-average basis varied from about 2 cm/year in Reaches 1 and 2 to over 15 cm/year in Reach 3 (not including the Upper Turning Basin). Reach 2 was separated into two sub-reaches based on differences in net sedimentation rates, with the spatially averaged rate in Reach 2A (RM 2.2 to RM 2.6) about 50% higher than the spatially averaged rate in Reach 2B (RM 2.6 to RM 4.0) (Figure 3-7). In general, spatially averaged net sedimentation rates were similar in the navigation channel and bench areas, with differences between the zones being less than a factor of 2 in a particular reach.

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[35] Annual time scales refer to time periods of 1 to 10 years, focusing on average or "typical" conditions for the sediment transport processes examined or discussed. Temporal variability in the processes exists, but conclusions or observations generally relate to long-term average conditions.

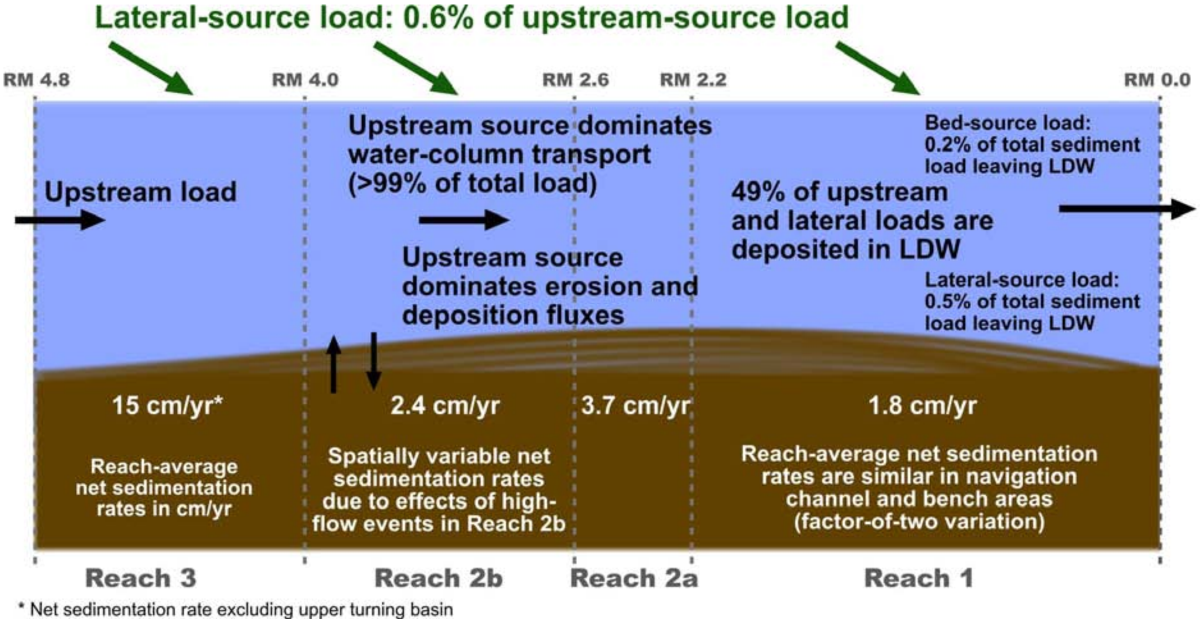

Figure 3-7. Schematic of net LDW sediment transport processes over a 30-year period

Three of the four questions that were established for the STM (see Section 3.2.1) were related to sediment bed stability. The questions and answers are provided below. The answers are based on STM results that represent LDW-wide conditions. Implications for specific areas with respect to remedial decisions will be considered in the FS using several lines of evidence.

- **What areas in the LDW are net depositional, net erosional, or in dynamic equilibrium?** Over the 30-year period, three grid cells in Reach 2 were estimated to be net erosional, but these locations should be considered to be in a state of approximate dynamic equilibrium because the total net erosion over the 30-yr period was estimated to be less than 1 cm. Other than these three locations, the sediment bed in the rest of the LDW was net depositional over the 30-year period, with average net sedimentation rates ranging from less than 0.1 cm/year to greater than 3 cm/yr.
- **What is the effect of high-flow events on episodic scour in otherwise net depositional areas?** Within Reach 1, negligible scour was estimated to occur during high-flow events in most of this reach, with only a minor effect on long-term deposition. A small area near RM 0.8 to RM 0.9 was estimated to experience bed scour during high-flow events, which tended to reduce long-term deposition and resulted in a relatively slow reduction in bed-source content. A small, localized area near RM 0.1 may also experience bed scour during high-flow events. Bed scour estimates were spatially variable in Reach 2,

and episodic erosion[36] during rare events tended to cause relatively low long-term net sedimentation rates at some locations. The relatively high net sedimentation rate estimates in Reach 3 (i.e., greater than 3 cm/yr) minimized the effects of bed scour during high-flow events over multi-year periods.

- What is the potential depth of scour during high-flow events in areas that are net depositional, net erosional, or in dynamic equilibrium? The depth of bed scour was dependent on river flow and tidal conditions during the high-flow event, as well as on the location in the LDW. For a 100-year high-flow event, approximately 20% of the LDW bed area was estimated to experience net erosion, with most of the erosion occurring in Reach 2. A large portion of the net erosion was estimated for the surface layer (i.e., 0 to 10 cm), with about 6% of the LDW bed area having net erosion depths greater than 10 cm. Maximum net erosion depth estimates during a 100-year high-flow event were about 21 cm, although such high net erosion occurred over only very small areas.

3.3.2 Changes in sediment bed composition

Based on the modeling, approximately 99% of the total external sediment particle load (by weight)[37] to the LDW comes from the upstream Green River (Figures 3-2 and 3-7). The annual average sediment load from lateral sources combined was estimated to be 0.6% of the annual average load from the Green River. Approximately 50% of the external sediment load, from upstream and lateral sources, was estimated to be deposited (or trapped) in the LDW (Figure 3-7). Of these deposited particles, approximately 50% are estimated to be medium-coarse sand; the other 50% of the deposited particles consist of fine sand (18% of the total deposited), medium-coarse silt (21% of the total deposited), and fine silt-clay (11% of the total deposited). The characteristics of the deposited particles vary by reach. Approximately 85% of the deposited particles in Reach 3 were estimated to be sand, whereas only 10% and 26% of the deposited particles were estimated to be sand in Reaches 1 and 2, respectively. TE (i.e., portion of incoming sediment load to each reach that is deposited within each reach) also varied among the three reaches as a result of differences in the hydrodynamic and sediment transport characteristics of these reaches. TE was highest for Reach 3 (36%) and lowest for Reach 2 (9%) (Figure 3-3).

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[36] Episodic erosion is bed scour that occurs during an episodic high-flow event or as a result of ship movement. The occurrence of episodic erosion at a particular location does not necessarily mean that an erosional environment exists at that location; a depositional or dynamic equilibrium environment can experience episodic erosion. During these events, current velocities are sufficiently fast to erode the bed at some locations. Generally, episodic erosion occurs over periods of hours to days.

[37] Particle loading should be distinguished from chemical loading, which may differ significantly, depending on the location. For example, chemical loads from lateral sources could have a significant influence on the sediment and water quality in localized areas, even though the relative magnitude of particle loading from those sources is very small compared with particle loading from upstream sources.

Of the sediment load leaving the LDW, sediment originating from the upstream source also represents over 99% of the total suspended solids (TSS) load transported downstream of the LDW (i.e., past RM 0.0), with sediment originating from lateral and bed sources composing about 0.5% and less than 0.2% of the TSS load exiting the LDW at RM 0.0, respectively (Figure 3-2). The contribution from lateral sources to the TSS load exiting the LDW at RM 0.0 was estimated to be about two to three times greater than the contribution from the bed-source sediment.

The bed-source content of the surface layer (0 to 10 cm) was estimated to decrease with time at an approximately exponential rate, primarily because of the deposition of upstream source sediment. The rate of decrease was spatially variable within the LDW as a result of variations in net sedimentation rate. According to net sedimentation rate calculations from the STM (Map 3-9) and Equation 3-5, approximately 90% of the total bed area in the LDW receives 10 cm of new sediment within 10 years or less. In other words, 90% of the LDW has estimated net sedimentation rates of at least 1.0 cm/year (i.e., 1.0 cm/year over a 10-year period equals an additional 10 cm of sediment) (Table 3-10). This sediment is mixed with the existing bedded sediment through various processes, including bioturbation and prop wash. In Reaches 2 and 3, elevated lateral-source contents (i.e., greater than 1%) in surface layer sediment were estimated to generally occur near the SD, CSO, and/or stream discharge points. The effects of lateral sources on surface layer composition were more widely distributed in Reach 1, with elevated lateral-source content values (greater than 1%) occurring over a large portion of this reach. At the end of a 30-year period, reach average values of lateral-source content in the surface layer of the bed were about 1 to 2% (Figure 3-8).

Table 3-10. Cumulative areal percentage of estimated net sedimentation rates

ESTIMATED NET SEDIMENTATION RATE (cm/yr)	PERCENTAGE (%)	CUMULATIVE PERCENTAGE (%)
≤ 0.5	4.7	4.7
> 0.5 and ≤ 1.0	5.9	10.6
> 1.0 and ≤ 1.5	23.5	34.1
> 1.5 and ≤ 2.0	16.1	50.2
> 2.0 and ≤ 2.5	12.4	62.6
> 2.5 and ≤ 3.0	7.0	69.6
> 3.0 and ≤ 4.0	7.6	77.2
> 4.0 and ≤ 6.0	9.3	86.5
> 6.0 and ≤ 10.0	4.3	90.8
> 10.0	9.2	100

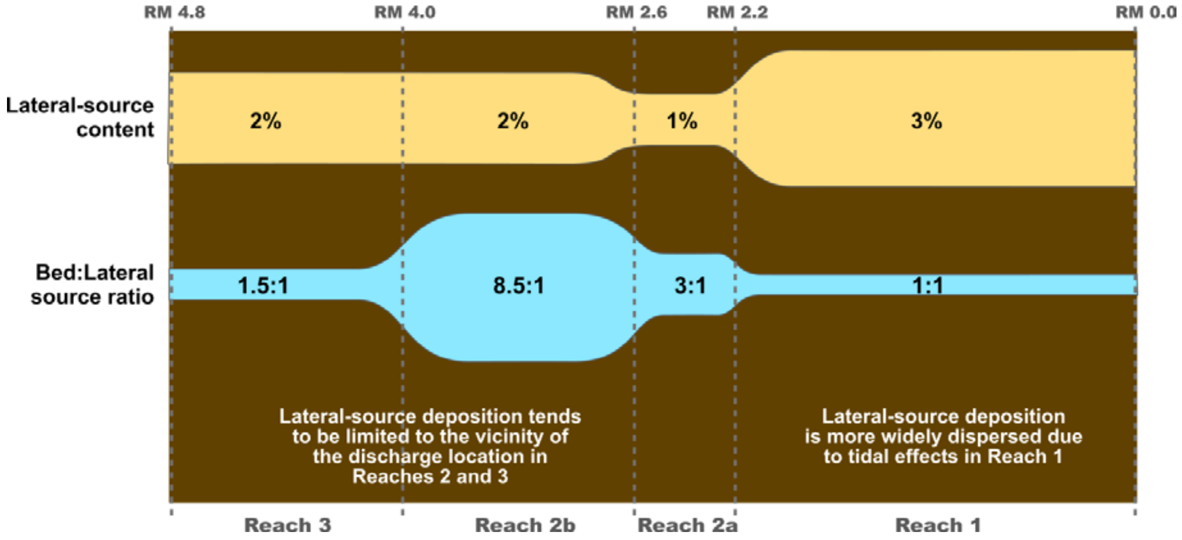

Figure 3-8. Estimated lateral-source content in surface layer (0 to 10 cm)

The fourth question that was established for the STM (see Section 3.2.1) concerns the change in the composition of the surface sediment layer over time as external sediment loads become incorporated into the sediment bed. The average bed-source content of the surface layer (i.e., 0 to 10 cm deep) sediment decreased by 97, 87, and 97% in Reaches 1, 2, and 3, respectively, during the 30-year period. In Reaches 2A and 2B, bed-source content decreased by 97 and 83%, respectively. For areas with net sedimentation rates less than about 0.2 to 0.3 cm/yr, bed-source content in the surface layer decreased at a relatively low rate. There are localized areas with little change in estimated original bed composition, where surface sediment concentrations exceeded the sediment quality standards (SQS) of the SMS, and net burial by sediment from the Green River is not occurring at significant rates. Many of these areas coincide with designated EAAs.

3.3.3 Model predictive capability and reliability

Calibration and validation of the STM, in conjunction with the spatial-scale and uncertainty analyses described in the STM report (QEA 2008), were used to evaluate the accuracy and reliability of the model. The 21-year simulation period used to calibrate and validate the STM was a strong test of the model's capabilities because of the wide range of tidal and river flow conditions during that period. Results of the calibration and validation exercises, as well as the spatial-scale and uncertainty analyses, indicate that the STM is able to adequately predict net sedimentation rates and bed composition in the navigation channel and bench areas at various spatial scales, which indicates that the model effectively simulates sediment transport processes in the LDW. Based on these results, the following conclusions concerning model reliability were developed:

- The STM may be used to refine, confirm, and validate the CSM.

- The analysis provides quantitative uncertainty estimates for STM predictions and CSM components.
- The STM provides a framework to support the evaluation of physical processes that could influence sediment and the effects of potential remedial actions in the LDW.
- Over small spatial-scales, on the order of a grid cell or several grid cells (94% of the grid cells are 1 ac or less), the STM provides one line of evidence, useful in conjunction with other information and data, to guide decision-making.
- The STM provides a reliable framework to extrapolate conditions in areas where no erosion and/or net sedimentation rate data are available.

The reliability of the STM makes it possible to use model results to support FS analyses at multiple spatial scales. The model provides a reliable framework for use as a diagnostic and prognostic tool to extrapolate information to areas in the LDW where no or minimal data are available for FS evaluation. However, the STM provides only one line of evidence for the FS analysis, which will rely on multiple lines of evidence to reach conclusions about the efficacy of one or more remedial alternative.

3.4 SUMMARY OF CONCEPTUAL SITE MODEL FOR SEDIMENT TRANSPORT

A range of empirical and modeling analyses was conducted by Windward and QEA (2008) and QEA (2008), with each analysis focusing on a specific component of sediment transport and bed stability in the LDW. The models developed in the STAR and STM provided several lines of evidence with similar conclusions; however, the uncertainty associated with the results and limitations in the models must be acknowledged. The qualitative conclusions derived from these analyses have a relatively low level of uncertainty, whereas the quantitative results have a higher level of uncertainty. The sensitivity analyses demonstrated that model results have an uncertainty level of a factor of 2 or less that is associated with uncertainty in model input parameters. This level of uncertainty is acceptable for the intended uses of the STM.

The results of the empirical analyses and modeling presented in Section 3.1 and the STM presented in Section 3.2 were used to refine the CSM presented in the STAR (in which the CSM was primarily based on the hydrodynamics and geomorphology), and even earlier versions presented in the Phase 1 RI (in which the CSM was primarily based on sediment properties). There have been step-wise refinements in the CSM developed over the past 4 yrs based on additional studies and modeling analyses that have been conducted for the RI/FS; the CSM may continue to be refined throughout the FS. The key concepts that define the revised CSM include the hydrodynamics of the saltwater estuary, sediment transport characteristics, erosion properties, and bed composition changes, each of which is summarized below.

3.4.1 Waterway-wide characteristics

- The entire LDW was estimated to be net depositional over an annual time scale.
- Net sedimentation rates were generally higher in the navigation channel than in the bench areas. For the navigation channel, the estimated net sedimentation rate decreased from the Upper Turning Basin (near RM 4.5) to downstream areas. For the bench areas, estimated net sedimentation rates were higher in Reaches 1, 2A, and 3 than in Reach 2B. Net sedimentation rate estimates tended to be lower in the intertidal areas than in the subtidal areas. Bed erosion is an episodic process that can be expected to be most pronounced during high-flow events. Episodic bed scour was estimated to occur to the greatest extent in Reach 2, was lower in Reach 3 than in Reach 2, and was minimal in Reach 1.[38] Net erosion is estimated to occur over about 20% or less of the LDW bed area during high-flow events, with most of the net erosion depth estimates being less than 10 cm (maximum of 21 cm or less).
- Ship-induced bed scour tends to behave as a mixing process for surficial sediment. In the scour model, the reworked surficial layer had an upper-bound average thickness of less than about 1 cm in the navigation channel and less than about 1 to 2 cm in the bench areas, with the frequency of such mixing being about 100 to 250 events per year. The scour model does not address localized scour around piers and berthing areas. The extent of this localized scour is unknown.

The CSM states that the LDW is net depositional over annual time scales, with a variable rate of net deposition (i.e., net sedimentation rate). The best estimate of the spatial distribution of net sedimentation rates in the LDW is presented on Map 3-9. Net depositional areas can be categorized as follows:

- **Lower net depositional areas**: areas with net sedimentation rates less than 0.5 cm/yr. In small, isolated areas within this category, the net sedimentation rate is minimal (e.g., less than 0.1 cm/yr) and the bed may approach a state of dynamic equilibrium (i.e., minimal changes in bed elevation over annual time scales).
- **Intermediate net depositional areas**: areas with net sedimentation rates ranging from 0.5 to 2.0 cm/yr.
- **Higher net depositional areas**: areas with net sedimentation rates greater than 2.0 cm/year and up to 140 cm/year in Reach 3.

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[38] Similar conclusions were presented in the STAR, but the reach designations were different in the STAR compared with the current CSM. For example, Reach 3 was defined as upstream of RM 3.0; it is now defined as upstream of RM 4.0.

The CSM is discussed below for each of the three LDW reaches separately, based on results of the various modeling exercises. Viewing these three reaches separately provides a more comprehensive understanding of sediment dynamics and bed stability within the LDW.

3.4.2 Reach 1: RM 0.0 to RM 2.2

Reach 1 is net depositional on annual time scales in both the navigation channel and the adjacent bench areas. Based on net sedimentation rates estimated by the model, the navigation channel is classified as intermediate and higher net depositional, with a small area near RM 0.8 to RM 0.9 with lower net deposition. The bench areas range from intermediate to higher net depositional, with two small areas classified as lower net depositional. With respect to episodic erosion, this reach is always within the saltwater wedge, even during a 100-year high-flow event. The permanent presence of the saltwater wedge serves as a protective barrier for the bed within this reach. Consequently, bed shear stresses (i.e., near-bed current velocities) are dominated by tidally driven flows, which are relatively low for all flow conditions. As a result, relatively low bed scour (less than 2 cm) is limited to a small area near RM 0.8 to RM 0.9. The potential for re-exposing buried sediments as a result of scour during high-flow events is minimal in this reach. Ship-induced mixing of the surficial bed layer potentially extends to average depths of about 1 to 2 cm in the bench areas and less than 1 cm in the navigation channel.

3.4.3 Reach 2: RM 2.2 to RM 4.0

Reach 2 is net depositional on annual time scales; however, estimated net sedimentation is spatially variable in this reach, ranging from lower to higher net depositional. Reach 2 includes the saltwater wedge toe during high-flow events; the saltwater wedge extends even farther upstream during average-flow conditions. More net erosion is estimated for this reach during high-flow events than for Reaches 1 and 3, but erosion is generally limited to the upper 10 cm of the sediment bed with maximum net erosion depths of approximately 21 cm or less in small areas. Ship-induced mixing of the surficial bed layer potentially extends to average depths of less than 1 cm in the bench areas and less than 0.1 cm in the navigation channel.

3.4.4 Reach 3: RM 4.0 to RM 4.8

Reach 3 is net depositional on annual time scales. The relatively high net sedimentation rates in this reach indicate that the navigation channel and bench areas are classified as higher net depositional. Modeling results indicate that episodic erosion may occur during high-flow events in Reach 3, but the areal extent of net erosion is significantly less than the areal extent of net erosion in Reach 2. Bed scour during high-flow events (i.e., return period of 2 yrs or greater) is generally limited to the upper 15 cm of the sediment bed, with maximum scour depths of 20 cm. Ship-induced mixing of the surficial bed layer potentially extends to average depths of less than 1 cm in the bench areas and less than 0.1 cm in the navigation channel.

4 Nature and Extent of Contamination

This section discusses the nature and extent of contamination in the LDW based on available data for sediment, tissue, and water. Section 4.1 discusses how data were selected for use in the RI. Section 4.2 presents summaries of chemical distributions in sediment, tissue, and water, with a focus on five chemicals or chemical groups (PCBs, arsenic, carcinogenic PAHs [cPAHs], dioxins and furans, and bis(2-ethylhexyl) phthalate [BEHP]),[39] which are discussed in separate subsections. Section 4.3 discusses total PCB concentrations in subsurface sediment in combination with information on sediment transport to determine whether vertical patterns of PCBs are consistent with the predictions of the STM and the CSM.

4.1 DATA SELECTION, SUITABILITY, AND REDUCTION

This section presents the data quality objectives (DQOs) for available tissue, sediment and water chemistry data for the LDW and describes how data were selected for use in the RI. It also describes how raw data from the laboratories were managed for use in the evaluation of the nature and extent of contamination.

4.1.1 Data quality objectives

DQOs were established to determine whether chemistry data were acceptable for all uses in the RI/FS. The DQO process used to identify acceptable datasets was reviewed and approved by EPA and Ecology (Windward 2001, 2005k). Table 4-1 lists the DQOs that must be satisfied for chemistry data to be considered acceptable for all uses in the RI/FS, categorizing them according to the level at which each DQO would be applied: event, station, sample, or result. A DQO applied at the result level could cause a result record to be qualified for a particular chemical but not for other chemicals analyzed during the same study.

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[39] PCBs, arsenic, cPAHs, and dioxins and furans were selected because they are risk driver chemicals for ecological receptors and/or humans, as described in Sections 5.5 and 6.5, respectively. BEHP was selected because it is a risk driver chemical for the benthic invertebrate community with the second highest number of SQS or CSL exceedances (after PCBs).

Table 4-1. Data quality objectives for chemistry data to be considered acceptable for all uses in the RI/FS

Level	Data Quality Objective
Event	Hard copy or original electronic copy of data report must be available.
	Field coordinates must be available.
	Data must have been collected since 1990.
	Data must have been collected using appropriate sampling methods.
	Existence and location of supporting documentation (i.e., analytical raw data, chain-of-custody forms, and sample handling descriptions) must be known.[a]
Station	Stations located within dredge prisms or remediated areas should be identified.
Sample	Sediment sample depth must be identified.
	Sample type must be clearly identified.
Result	Data validation qualifiers must be present, or derivable from laboratory qualifiers or QA information, and must be applied in a manner consistent with EPA functional guidelines (EPA 1999c, 2002e).[a]
	Each result must have a laboratory-generated form (usually referred to as a Form 1).[a]
	For non-detects, reporting limits and appropriate qualifiers must be given.
	Calculated values must be recalculated.
	Analytical methods must be identified.

[a] The existence of supporting documentation, the availability of Form 1s, and a review of the presence and application of data validation qualifiers were not considered necessary if the data were previously approved by EPA for all uses in the RCRA or CERCLA programs.

CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
EPA – US Environmental Protection Agency
FS – feasibility study
RCRA – Resource Conservation and Recovery Act
RI – remedial investigation

EPA has not established definitive guidelines specifying the level of data validation required for Superfund investigations. However, EPA Order 5360.1 and Office of Solid Waste and Emergency Response (OSWER) Directive 9355.9-01 (EPA 1993a) requires that environmental measurements be of known quality, verifiable, and defensible. The Office of the Inspector General (OIG) concluded in an audit of Region 9 Superfund sites (EPA 1995) that data used for cleanup decision-making should be validated using EPA functional guidelines (EPA 1999c, 2002e). According to these guidelines, two different levels of data validation are generally recognized for chemistry data. A summary data validation, referred to as QA1, represents a lower level of effort compared with a full validation, referred to as QA2. The elements of summary and full data validations for environmental chemistry data are presented in Table 4-2 (EPA 1999c, 2002e).

Table 4-2. Elements of summary and full data validations for environmental chemistry data

Element	Applicable Analyses	Summary Data Validation (QA1)	Full Data Validation (QA2)
Quality control analysis frequencies	all	X	X
Analysis holding times	all	X	X
Instrument performance check	organic compounds, ICP-MS metals		X
Initial instrument calibration	all		X
Continuing instrument calibration	all		X
Laboratory blanks	all	X	X
ICP interference check sample	metals		X
System monitoring compounds (surrogates)	organic compounds	X	X
Matrix spikes/matrix spike duplicates	all	X	X
Laboratory control samples	all	X	X
ICP serial dilution	metals		X
Field QA/QC (field blanks, field duplicates)	all	X	X
Internal standards	VOCs, SVOCs, ICP-MS metals		X
Pesticide cleanup checks	pesticides/PCBs		X
Target compound identification and quantitation (requires verification of reported results with raw data)	organic compounds		X
RLs	all	X	X

ICP-MS – inductively coupled plasma-mass spectrometry
PCB – polychlorinated biphenyl
QA/QC – quality assurance/quality control
RL – reporting limit
SVOC – semivolatile organic compound
VOC – volatile organic compound

Data of acceptable quality may still be associated with uncertainty in the RI. For example, a chemical not detected in a sample may actually be present, but its concentration below the reporting limit (RL) is unknown. This uncertainty applies to all cases in which chemicals are reported as undetected; the magnitude of this uncertainty increases with increasing RLs. None of the sampling events evaluated for inclusion in the RI were excluded in their entirety because of elevated RLs. The uncertainties associated with data quality and that are relevant to conclusions of the risk assessments are discussed in both the ERA (Appendix A) and HHRA (Appendix B).

4.1.2 Data selection

Many environmental investigations conducted within the LDW have included the collection of samples of surface sediment, subsurface sediment, fish or shellfish tissue, surface water, sediment porewater, or seep water for chemical analysis. This section describes the datasets selected for defining the nature and extent of contamination.

4.1.2.1 Surface sediment

Surface sediment samples have been collected during numerous sampling events since 1990 (Table 4-3). A baseline surface sediment chemistry dataset was compiled to characterize sediment from the LDW prior to any cleanup actions that occurred after the start of the RI in 2001 (i.e., excluding data collected after cleanup actions at the Duwamish/Diagonal EAA in 2003/2004 and 2005 and at the BDC south storm drain outfall at the Norfolk EAA in 2003[40]). This baseline dataset, which was used in the ERA and HHRA, included surface sediment data collected during the first two rounds of sampling conducted for the RI. A third round of surface sediment data was collected in 2006; these data were combined with the baseline dataset used in the ERA and HHRA to characterize the nature and extent of contamination. This combined baseline/Round 3 dataset is referred to in the remainder of this document as the RI baseline dataset. The criteria for including or excluding surface sediment data in the RI baseline dataset were determined in agreement with EPA and Ecology, as follows (Windward 2006d):

- Surface sediment data were excluded if they were collected before 1991 or after 2005 if they were not sampled as part of the RI. Surface sediment data collected as part of the RI in 2006 were included in the baseline dataset.
- Surface sediment data were excluded if they did not meet the DQOs, as described in Section 4.1.1.
- Surface sediment samples were excluded if the sampled areas were dredged as part of maintenance activities after the sample had been collected.
- Older data were excluded if a sediment sampling location was resampled at a later date within 10 ft of the original location, either as part of a monitoring program or to characterize temporal trends.[41]
- Surface sediment samples were excluded if they were collected after removal actions at two EAAs (i.e., Duwamish/Diagonal and BDC south storm drain outfall near RM 3.9, so that the dataset would represent baseline (pre-early action) conditions.[42]

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[40] Data for these areas following these actions are presented in Sections 4.2.2.1 and Appendix I.4.

[41] If data for an older sample were trumped by a newer sample, data for the older sample were not retained in the RI baseline dataset. Appendix E contains a summary of the data that were omitted as a result of this process (see Table E.1-4). The FS surface sediment dataset will include data for chemicals analyzed in older samples that were not analyzed in newer samples; these data will be included in FS analyses and maps.

[42] There is no EPA policy or guidance on whether baseline risk assessments should take into account the risk reduction achieved by removal actions that occur during the RI/FS. The risk reduction achieved

- Post-cleanup sediment samples near the Duwamish/Diagonal EAA were included only if they had been collected outside of a 200-ft perimeter of the EAA after the removal action (completed in 2004) because samples within 200 ft could have been influenced by the action.
- Sediment samples were categorized as surface sediment samples if they were collected to a depth of 15 cm or less. At 24 locations in the Norfolk EAA, samples were collected from the 0-to-2-cm depth in addition to the 0-to-10-cm depth on either the same date or at a later date. Only data from the 0-to-10-cm depth were included in the RI baseline dataset.

Table 4-3. Selection of surface sediment samples for the RI baseline dataset

FINAL DETERMINATION	NUMBER OF SAMPLES
Included in RI baseline surface sediment dataset.	1,484[a]
Excluded because of subsequent dredging of those sediments.	58
Excluded because of more recent sampling outside of monitoring program.	95
Excluded because of more recent sampling as part of monitoring program.	30
Excluded because of data quality considerations.	23
Excluded because 0-to-10-cm samples were preferred over 0-to-2-cm samples at these locations.	24
Excluded because post-removal sampling was not appropriate for the RI baseline surface sediment dataset at this location (Duwamish/Diagonal).	50
Total number of surface sediment samples	**1,764**

[a] Samples were collected from 1,365 discrete locations.

RI – remedial investigation

Both intertidal and subtidal sediment chemistry data were represented in the RI baseline dataset. The boundary used to differentiate between intertidal and subtidal areas was delineated by the US Fish and Wildlife Service (USFWS) as the shoreline (i.e., land/water interface) at low tide (estimated water level of -2 ft MLLW) using aerial photographs taken in 1999 (2000c). The 1,484 surface sediment samples in the RI baseline dataset were collected from 1,365 discrete locations; 593 of these locations are intertidal and 772 are subtidal (Maps 4-1 and 4-2).

Data quality reviews were conducted on each dataset that met the above criteria and was thus considered for use in the RI/FS. These reviews culminated in a determination of whether an individual dataset was considered suitable for all uses in the RI/FS (Windward 2005k, l, 2007a, e). Summaries of these data quality reviews are presented in Appendix E, Section E.2.

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at these two areas will be evaluated in the FS. Surface sediment data collected after dredging or other remedial activities at the Duwamish/Diagonal and BDC south storm drain areas are described in Sections 4.2.2.1, 10.3.1, and 10.3.15.

Appendix E, Section E.1, presents a more detailed description of the process for selecting data collected from dredged areas, from resampled locations less than 10 ft apart, and from the Duwamish/Diagonal EAA and the BDC south storm drain area. Table E.2-3 in Appendix E, Section E.2, identifies the surface sediment samples that were excluded from the RI baseline surface sediment dataset and the rationale for their exclusion. Approximately 85% of the surface sediment samples (1,484 of 1,764) were included in the RI baseline dataset, as presented in Table 4-3.

The surface sediment sampling events included in the RI baseline dataset are listed in Table 4-4. Additional surface sediment sampling events conducted in the LDW since 1990, but not included in the baseline dataset for one of the reasons presented in Table 4-3, are listed in Table 4-5. These data are discussed from a temporal perspective in Sections 4.2.3.1 (Duwamish/Diagonal) and Appendix I, Sections I.4.1 and I.4.8 (Duwamish/Diagonal and Norfolk). The locations of the RI baseline surface sediment samples collected during site-specific or LDW-wide studies are shown on Maps 4-1 and 4-2, respectively, and sampling locations with their sample identifications (IDs) are shown on Maps 4-3 and 4-4a through 4-4j.

Table 4-4. Summary of sampling events included in the RI baseline surface sediment dataset

SAMPLING EVENT	EVENT CODE	YEAR OF SAMPLING	CHEMICALS	NUMBER OF SAMPLING LOCATIONS[a]	SOURCE
LDW RI: surface sediment sampling for chemical analyses	LDWRI-SurfaceSediment Round3	2006	metals, SVOCs, PCB Aroclors, dioxins and furans on subset of samples, butyltins, conventionals[b]	44	Windward (2007c)
LDW RI: chemical analyses of benthic invertebrate and clam tissue samples and co-located sediment samples	LDWRI-Benthic	2005	metals, SVOCs, PCB Aroclors, selected PCB congeners on subset of samples, butyltins, conventionals[b]	35	Windward (2005a)
LDW RI: surface sediment sampling for chemical analyses and toxicity testing	LDWRI-SurfaceSediment Round1 & Round2	2005	metals, SVOCs, PCB Aroclors, selected PCB congeners and dioxins and furans on subset of samples, butyltins, conventionals[b]	160	Windward (2005c, d)
Boyer Towing dock replacement	Boyer Towing	2004	metals, SVOCs, PCB Aroclors, TBT, conventionals[b]	3	WR Consulting (2004)
Slip 4 EAA site characterization	Slip4-EarlyAction	2004	PCB Aroclors, mercury	30	Integral (2004)
Rhône-Poulenc surface/subsurface sampling	RhônePoulenc 2004	2004	metals, organochlorine pesticides, SVOCs, conventionals[b]	21	EPA (2005d)
Triad approach to characterize PCB in a Washington riverine sediment site (USACE)	Jorgensen August 2004	2004	metals, PCB Aroclors, SVOCs, conventionals[b]	43	Wakeman (2005)
Duwamish/Diagonal perimeter monitoring – pre-dredge	DuwDiag-October2003	2003	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[b]	12	King County et al. (2005b)

SAMPLING EVENT	EVENT CODE	YEAR OF SAMPLING	CHEMICALS	NUMBER OF SAMPLING LOCATIONS[a]	SOURCE
Terminal 117 EAA site characterization	T117 Boundary Definition	2003-2004	PCB Aroclors; metals, TBT and SVOCs on selected samples, conventionals[b]	54	Windward et al. (2004a, b)
Boeing Plant 2 transformer investigation – Phase 1	Plant 2-Transformer Phase1	2003	PCB Aroclors, conventionals[b]	6[c]	Floyd Snider McCarthy (2004)
Norfolk CSO (Duwamish River) sediment cap recontamination – Phase I investigation	Ecology-Norfolk	2002	PCB Aroclors, conventionals[b]	17	Ecology (2003)
Norfolk CSO sediment remediation project 5-year monitoring program: Annual monitoring report – year 3, April 2002	Norfolk-monit5	2002	metals, PCB Aroclors, SVOCs, conventionals[b]	1	King County (2002a)
Norfolk CSO 5-year monitoring program, year 2, April 2001	Norfolk-monit4	2001	metals, PCB Aroclors, SVOCs, conventionals[b]	1	King County (2001a)
Norfolk CSO 5-year monitoring program – 12-month post-construction	Norfolk-monit3	2000	metals, PCB Aroclors, SVOCs, conventionals[b]	1	King County (2000c)
Norfolk CSO 5-year monitoring program – supplemental nearshore sampling	Norfolk-monit2b	2000	metals, PCB Aroclors, SVOCs, conventionals[b]	3	King County (2000b)
Outfall and nearshore sediment sampling report, Duwamish facility	James Hardie Outfall	2000	metals, PCB Aroclors, SVOCs, conventionals[b]	9	Weston (2000)
Norfolk CSO 5-year monitoring program – 6-month post-construction	Norfolk-monit2a	1999	metals, PCB Aroclors, SVOCs, conventionals[b]	2	King County (2000d)
Norfolk CSO 5-year monitoring program – post-backfill	Norfolk-monit1	1999	metals, PCB Aroclors, SVOCs, conventionals[b]	2	King County (1999f)
EPA site inspection: Lower Duwamish River	EPA SI	1998	metals, PCB Aroclors, selected PCB congeners, and SVOCs; organochlorine pesticides, dioxins and furans, TBT, and VOCs on subset of samples; conventionals[b]	251	Weston (1999a)
King County CSO water quality assessment for the Duwamish River and Elliott Bay	KC WQA	1997	metals, PCB Aroclors, SVOCs, tetrabutyltin, conventionals[b]	14	King County (1999e)
Duwamish Waterway Phase 1 site characterization	Boeing SiteChar	1997	metals, PCB Aroclors, SVOCs, conventionals[b]	79[d]	Exponent (1998)
Duwamish Waterway sediment characterization study	NOAA SiteChar	1997	total PCBs, selected PCB congeners, total polychlorinated terphenyls, conventionals[b]	299	NOAA (1997, 1998)
Seaboard Lumber Phase 2 site investigation	Seaboard-Ph2	1996	metals, PCB Aroclors, SVOCs, conventionals[b]	20	Herrera (1997)
Rhône-Poulenc seep sampling	Rhône-Poulenc RFI-3	1996	metals, phenols	14	Rhône-Poulenc (1996)

SAMPLING EVENT	EVENT CODE	YEAR OF SAMPLING	CHEMICALS	NUMBER OF SAMPLING LOCATIONS[a]	SOURCE
RCRA facility investigation Duwamish Waterway sediment investigation, Plant 2 – Phase 2b	Plant 2 RFI-2b	1996	metals, PCB Aroclors, phthalates, conventionals[b]	36	Weston (1998)
Duwamish/Diagonal cleanup study – Phase 2	Duw/Diag-2	1996	metals, PCB Aroclors, SVOCs, conventionals[b]	10	King County (2000a)
Duwamish/Diagonal cleanup study – Phase 1.5	Duw/Diag-1.5	1995	metals, PCB Aroclors, SVOCs, conventionals[b]	9	King County (2000a)
Norfolk CSO sediment cleanup study – Phase 3	Norfolk-cleanup3	1995	PCB Aroclors, SVOCs, conventionals[b]	12	King County (1996)
Norfolk CSO sediment cleanup study – Phase 2	Norfolk-cleanup2	1995	metals, organochlorine pesticides, PCB Aroclors, selected PCB congeners, SVOCs, conventionals[b]	2	King County (1996)
RCRA facility investigation, Duwamish Waterway sediment investigation, Plant 2 – Phase 2a	Plant 2 RFI-2a	1995	metals, PCB Aroclors SVOCs, conventionals[b]	54	Weston (1998)
RCRA facility investigation, Duwamish Waterway sediment investigation, Plant 2 – Phase 1	Plant 2 RFI-1	1995	metals, PCB Aroclors, TPH, SVOCs, VOCs, conventionals[b]	66	Weston (1998)
Duwamish/Diagonal cleanup study – Phase 1	Duw/Diag-1	1994	metals, organochlorine pesticides, PCB Aroclors, SVOCs, conventionals[b]	31	King County (2005a)
Norfolk CSO sediment cleanup study – Phase 1	Norfolk-cleanup1	1994	metals, organochlorine pesticides, SVOCs, PCB Aroclors, conventionals[b]	13	King County (1996)
Rhône-Poulenc RCRA facility investigation for the Marginal Way facility – Round 2	Rhône-Poulenc RFI-2	1994	SVOCs, conventionals[b]	6	Rhône-Poulenc (1995)
Results of sampling and analysis, sediment monitoring plan, Duwamish Shipyard, Inc.	Duwamish Shipyard	1993	metals, SVOCs, TBT, conventionals[b]	1	Hart Crowser (1993)
Harbor Island remedial investigation	Harbor Island RI	1991	metals, organochlorine pesticides, PCB Aroclors, SVOCs, VOCs, TPH, TBT, conventionals[b]	9	Weston (1993b)

[a] Samples are surface sediment grab samples from 0 to 15 cm unless otherwise noted.

[b] Conventionals may include all or some of the following parameters: grain size, pH, total organic carbon, and total solids.

[c] Five samples were collected from 0 to 5 cm. The top interval of 0 to 15 cm from a subsurface sediment core was also included in the RI baseline surface sediment chemistry dataset.

[d] Sample total does not include three reference samples that were collected upstream of the study area.

CSO – combined sewer overflow
EAA – early action area
EPA – US Environmental Protection Agency
LDW – Lower Duwamish Waterway
NOAA – National Oceanic and Atmospheric Administration
PCB – polychlorinated biphenyl
RCRA – Resource Conservation and Recovery Act
RFI – RCRA facility investigation
RI – remedial investigation
SI – site inspection
SVOC – semivolatile organic compound
TBT – tributyltin
TPH – total petroleum hydrocarbon
USACE – US Army Corps of Engineers
VOC – volatile organic compound
WQA – water quality assessment

Table 4-5. Summary of sampling events not included in the RI baseline surface sediment dataset

SAMPLING EVENT	EVENT CODE	YEAR	CHEMICALS	NUMBER OF SAMPLING LOCATIONS[a]	REFERENCE
BDC 2006 annual sampling of south storm drain system – year 3	Boeing Developmental Center-2007	2007	PCB Aroclors, conventionals[b]	3	CALIBRE (2007)
Duwamish Diagonal March 2007 cap monitoring – year 3	DuwDiagonal April2007	2007	metals, SVOCs, PCB Aroclors, organochlorine pesticides, conventionals[b]	23	King County (2007c, d, e)
Duwamish Diagonal March 2006 cap monitoring – year 2, perimeter sediment characterization and enhanced natural recovery cap sediment characterization – year 1	DuwDiagMarch 2006	2006	metals, SVOCs, PCB Aroclors, organochlorine pesticides, conventionals[b]	23	King County (2006d, e, 2007b)
Duwamish Diagonal January-February 2005 post-dredge perimeter – before thin-layer cap placement	DuwDiagJan 2005	2005	metals, SVOCs, PCB Aroclors, organochlorine pesticides, conventionals[b]	20	King County (2005g)
Duwamish Diagonal March 2005 post-dredge perimeter – after thin-layer cap placement	DuwDiagMarch 2005	2005	metals, SVOCs, PCB Aroclors, organochlorine pesticides, conventionals[b]	7	King County (2005f)
Duwamish Diagonal April 2005 baseline cap monitoring – year 1	DuwDiagApril 2005	2005	metals, SVOCs, PCB Aroclors, organochlorine pesticides, conventionals[b]	6	King County (2005e)
Duwamish Diagonal August 2005 baseline cap monitoring – year 1	DuwDiagonal August 2005	2005	metals, SVOCs, PCB Aroclors, organochlorine pesticides, conventionals[b]	1	King County (2005h)
Boeing Developmental Center 2005 annual sampling of south storm drain system – year 2	Boeing Developmental Center-2005	2005	PCB Aroclors, conventionals[b]	3	CALIBRE (2006)
Boeing Developmental Center 2004 annual sampling of south storm drain system – year 1	Boeing Developmental Center-2004	2004	PCB Aroclors, conventionals[b]	3	Calibre (2005)
Duwamish Diagonal June 2004 baseline cap monitoring – year 0 (post-cap placement)	DuwDiagJune 2004	2004	metals, PCB Aroclors, SVOCs, conventionals[b]	7	King County (2005i)
Duwamish Diagonal March 2004 post-dredge perimeter sediment characterization	DuwDiagonal-March2004	2004	metals, SVOCs, PCB Aroclors, organochlorine pesticides, conventionals[b]	12	King County (2004b)
Norfolk CSO sediment remediation project 5-year monitoring program; annual monitoring report – year 5, April 2004	Norfolk-monit7	2004	metals, PCB Aroclors, SVOCs, conventionals[b]	4[c]	King County (2005d)

SAMPLING EVENT	EVENT CODE	YEAR	CHEMICALS	NUMBER OF SAMPLING LOCATIONS[a]	REFERENCE
Norfolk CSO sediment remediation project five-year monitoring program: Annual monitoring report – year 4, April 2003.	Norfolk-monit6	2003	metals, PCB Aroclors, SVOCs, conventionals[b]	4[c]	King County (2003)

[a] Samples are surface sediment grab samples from 0 to 15 cm unless otherwise noted.

[b] Conventionals may include all or some of the following parameters: grain size, pH, total organic carbon, and total solids.

[c] These locations were resampled during the Round 3 RI surface sediment sampling in 2006; thus the data from 2003 and 2004 were superseded by the more recent data from 2006.

BDC – Boeing Developmental Center
EAA – early action area
PCB – polychlorinated biphenyl
RI – remedial investigation
SVOC – semivolatile organic compound
TOC – total organic carbon
VOC – volatile organic compound

The baseline surface sediment dataset used in both the ERA (Appendix A) and HHRA (Appendix B) did not include data collected during the Round 3 RI sampling event, and therefore contained fewer data. Round 3 sampling was conducted in October 2006, and results were not available for inclusion in the draft ERA or the draft HHRA, which were submitted to EPA and Ecology in August 2006 and September 2006, respectively. Prior to Round 3, the baseline dataset included a total of 1,446 surface sediment samples collected from 1,329 discrete locations throughout the LDW. Round 3 included the collection of 47 additional surface sediment samples from 44 discrete locations. Appendix C presents an evaluation of whether the risk assessment conclusions would have been different if Round 3 surface sediment data had been included.

4.1.2.2 Subsurface sediment

Numerous subsurface sediment sampling events have been conducted since 1990.[43] All subsurface sediment sampling events included in the RI dataset are summarized in Table 4-6 (for areas that have been dredged) and in Table 4-7 (for areas that have not been dredged) and the locations of subsurface samples collected during site-specific and LDW-wide studies are shown on Maps 4-5 and 4-6, respectively. Sampling locations, including the sample IDs, are presented on Maps 4-7a through 4-7d. Sampling locations with data for specific 2-ft core intervals are shown on Maps 4-8a through 4-8c.

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[43] Since 1990, no subsurface sediment datasets have been excluded from the RI dataset because of data quality issues.

Table 4-6. Summary of subsurface sediment sampling events included in the RI dataset in areas that have been dredged

SAMPLING EVENT	EVENT CODE	YEAR OF SAMPLING	CHEMICALS	SAMPLE SUMMARY	SAMPLE LOCATION STATUS	REFERENCE
PSDDA characterization at the Lehigh Northwest Duwamish Waterway facility	LehighNW	2004	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	3 samples (impact corer) from 4 locations; 2 from 0 to 120 cm, 1 from 120 to 150 cm below mudline	Area was dredged in 2004.	MCS (2004d)
Sediment characterization results for the Duwamish River navigational channel turning basin	Turning-basin	2003	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	5 samples (vibracorer) from 5 locations; collected to depths of 144 to 390 cm	Area was most recently dredged in 2004.	Anchor (2003)
Glacier Northwest cement terminal	Glacier NW	2002	metals, PCB Aroclors, organochlorine pesticides, SVOCs, TBT, conventionals[a]	9 samples (vibracorer), each made from 3 separate cores collected to 90 to 150 cm below mudline	Area was dredged in 2005.	Pacific International Engineering (2002)
Dredge material characterization, Duwamish Yacht Club	Duwam Yacht Club	1999	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, TBT, conventionals[a]	6 samples (vibracorer) from 6 locations, each made from 2 separate cores collected from 50 to 65 cm below mudline	Area was dredged in 1999.	Hart Crowser (1999)
Sediment sampling and analysis, James Hardie Gypsum, Inc. – Round 1	Hardie Gypsum-1	1999	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	5 samples (vibracorer) made from single cores down to 120 cm below mudline	Area was dredged in 1999.	Spearman (1999)
Sediment sampling and analysis, James Hardie Gypsum, Inc. – Round 2	Hardie Gypsum-2	1999	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	9 samples (vibracorer) made from single cores down to 90 cm below mudline	Area was dredged in 1999.	Spearman (1999)
EPA site inspection: Lower Duwamish River	EPA SI	1998	metals, PCB Aroclors and selected congeners, organochlorine pesticides, SVOCs, VOCs, TBT, conventionals[a]	33 samples (vibracorer) from 17 locations; 1 to 2 samples collected at each location, from 0 to 60 and 60 to 120 cm below mudline	The area at RM 0.5 (DR008) was dredged in 2003/2004.	Weston (1999a)
Dredge material characterization Hurlen Construction Company & Boyer Alaska Barge Lines berthing areas	Hurlen-Boyer	1998	metals, PCB Aroclors, organochlorine pesticides, SVOCs, TBT, TPH, conventionals[a]	6 samples (vibracorer) from 2 locations at Boyer and 4 locations at Hurlen, each made from 2 separate cores collected from 60 to 120 cm below mudline	Area was dredged in 1998.	Hart Crowser (1998)

Table 4-6, cont. Summary of subsurface sediment sampling events included in the RI baseline dataset in areas that have been dredged

SAMPLING EVENT	EVENT CODE	YEAR OF SAMPLING	CHEMICALS	SAMPLE SUMMARY	SAMPLE LOCATION STATUS	REFERENCE
PSDDA sediment characterization of Duwamish River navigation channel: FY1999 operations and maintenance dredging data report	PSDDA98	1998	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	10 samples (vibracorer) from 12 locations; 7 samples from 0 to 60 to 90 cm, each from single location; 3 samples from 2 or 3 locations (0 to 60 cm, 0 to 120 cm, and 120 to 360 cm below mudline)	Area was most recently dredged in 1999.	Striplin (1998)
Duwamish/Diagonal cleanup study – Phase 2	Duw/Diag-2	1996	metals, PCB Aroclors, SVOCs, TPH, conventionals[a]	37 samples (vibracorer) from 14 locations; 1 to 6 samples per core, up to 270 cm below mudline	Area was dredged in 2003/2004.	King County (2000a)
1996 USACE Duwamish O&M	PSDDA96	1996	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	4 samples (vibracorer) from 4 locations; collected to a depth of 120 cm below mudline	Area was most recently dredged in 1999.	Striplin (1996)
Proposed dredging of Slip No. 4, Duwamish River, Seattle, WA	Slip4-Crowley	1996	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, TBT, conventionals[a]	4 samples (vibracorer) composited from sediment at 9 locations; collected to a depth of 70 to 130 cm below mudline	Area was dredged in 1996.	PTI (1996)
Lone Star Northwest and James Hardie Gypsum – Kaiser dock upgrade	Lone Star-Hardie Gypsum	1995	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	5 samples (vibracorer) from 4 locations; 4 collected to a depth of 120 to 150 cm, 1 at 120 to 360 cm below mudline	Area was dredged in 1995.	Hartman (1995)
Norfolk CSO sediment cleanup study – Phase 2	Norfolk-cleanup2	1995	metals, PCB Aroclors and selected congeners, organochlorine pesticides, SVOCs, VOCs, TPH, conventionals[a]	27 samples (vibracorer) from 3 locations; collected at 30- or 60-cm intervals up to 180 cm below mudline	Area was dredged and capped in 1999.	King County (1996)
Duwamish/Diagonal cleanup study – Phase 1	Duw/Diag-1	1994	metals, PCB Aroclors, organochlorine pesticides, SVOCs, TBT, conventionals[a]	12 samples (vibracorer) from 2 locations; collected at 15- to 30-cm intervals down to 150 cm below mudline	Area was dredged in 2003/2004.	King County (2005a)
Norfolk CSO sediment cleanup study – Phase 1	Norfolk-cleanup1	1994	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	3 samples from 1 location; collected at 15 to 30, 30 to 45, and 45 to 60 cm below mudline	Area was dredged and capped in 1999.	King County (1996)
Lone Star Northwest – West Terminal USACE – Seattle, WA	Lone Star 92	1992	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	1 sample (vibracorer) made from 2 separate cores collected to 120 cm below mudline	Area was dredged in 1995.	Hartman (1992)
Sediment sampling and analysis, South Park Marina, Duwamish Waterway, Seattle, WA	South Park Marina	1991	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	2 samples (vibracorer), each made from 2 separate cores collected to 120 cm below mudline	Area was dredged in 1992.	Spearman (1991b)

Table 4-6, cont. Summary of subsurface sediment sampling events included in the RI baseline dataset in areas that have been dredged

Note: For reference, 1 ft is equal to 30 cm.

[a] Conventionals may include all or some of the following parameters: grain size, pH, total organic carbon, and total solids.

CSO – combined sewer overflow
EPA – Environmental Protection Agency
PCB – polychlorinated biphenyl
PSDDA – Puget Sound Dredged Disposal Analysis
RM – river mile
SVOC – semivolatile organic compound
TBT – tributyltin
TPH – total petroleum hydrocarbons
USACE – US Army Corps of Engineers
VOC – volatile organic compound

Table 4-7. Summary of subsurface sediment sampling events included in the RI dataset in areas that have not been dredged

SAMPLING EVENT	EVENT CODE	YEAR OF SAMPLING	CHEMICALS	SAMPLE SUMMARY	SAMPLE LOCATION STATUS	REFERENCE
LDW RI: Subsurface sediment sampling for chemical analyses	LDW Subsurface Sediment 2006	2006	metals, PCB Aroclors, organochlorine pesticides, SVOCs, dioxins and furans, conventionals[a]	213 samples (impact corer or vibracorer) from 56 locations at 1- or 2-ft intervals to depths ranging from 6 to 12 ft below mudline ; 62 samples (impact corer or vibracorer) from 9 locations at 0.5-ft intervals to depths ranging from 2 to 4.5 ft below mudline	Area has not been dredged or remediated.	Windward and RETEC (2007)
Rhône-Poulenc (Rhodia) sediment and porewater investigation	Rhône-Poulenc AugSep 2004	2004	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	11 subtidal locations (push cores to a depth of 30 cm below mudline) and 13 intertidal locations (clam gun cores to a depth of 1 m below mudline); 2 samples (0 to 10 cm and > 10 cm) from each subtidal and intertidal location	Area has not been dredged or remediated.	EPA (2005d)
Additional vertical characterization, Duwamish Sediment Other Area	DSOAvert char3	2004	PCB Aroclors, conventionals[a]	7 samples (impact corer) from 5 locations; 1 to 3 samples from each location from 60 to 180 cm below mudline	Area has not been dredged or remediated.	MCS (2004b)
Triad approach (immunoassay as a real-time measure) to characterize PCBs in a Washington riverine sediment site	Jorgensen August 2004	2004	metals, PCB Aroclors, conventionals[a]	51 samples (impact corer) from 16 locations; 1 to 5 samples per location from 45 to 180 cm below mudline	Area has not been dredged or remediated.	unpublished data from USACE
Duwamish Sediment Other Area upriver (Area 1) sediment characterization	Jorgensen April 2004	2004	metals, PCB Aroclors, conventionals[a]	75 samples (impact corer) from 22 locations; 2 to 9 samples per location from 0 to 265 cm below mudline	Area has not been dredged or remediated.	MCS (2004c)
Boyer Towing dock replacement	Boyer Towing	2004	metals, PCB Aroclors, SVOCs, conventionals[a]	3 samples (push core) from 3 locations; 30 to 60 cm below mudline	Area has not been dredged.	WR Consulting (2004)
Additional vertical characterization, Duwamish Sediment Other Area	DSOAvert char2	2004	PCB Aroclors, conventionals[a]	28 samples (impact corer) from 15 locations; 1 to 3 samples from each location from 60 to 144 cm below mudline	Area has not been dredged or remediated.	MCS (2004a)
Slip 4 EAA site characterization	Slip4-EarlyAction	2004	PCB Aroclors, mercury, conventionals[a]	57 samples (vibracorer) from 11 locations; 4 to 6 samples taken at each location to a depth of 360 cm below mudline	Area has not been dredged or remediated.	Integral (2004)
Terminal 117 (T-117) EAA site characterization	T-117 Boundary Definition	2003-2004	metals, PCB Aroclors; SVOCs (selected samples), conventionals[a]	107 samples (vibracorer) from 25 locations, 3 to 6 samples collected at each core location to a depth of 300 cm	Area has not been dredged or remediated.	Windward (2004a, b)

Table 4-7, cont. Summary of subsurface sediment sampling events included in the RI dataset in areas that have not been dredged

SAMPLING EVENT	EVENT CODE	YEAR OF SAMPLING	CHEMICALS	SAMPLE SUMMARY	SAMPLE LOCATION STATUS	REFERENCE
Preliminary site investigation for the South Park Bridge Project	SouthPark Bridge	2003	metals, TBT, VOCs, SVOCs, pesticides, PCB Aroclors, conventionals[a]	11 samples (mud-rotary drill unit) from 2 locations; 5 to 6 samples from each location at depths from 2.5 to 100 ft below mudline	Area has not been dredged.	Wilbur Consulting (2004)
Boeing Plant 2 transformer investigation – Phase 1	Plant 2-Transformer Phase1	2003	PCB Aroclors, conventionals[a]	46 samples (impact corer) from 13 locations; 3 to 5 samples at each location from 0 to 240 cm below mudline	Area has not been dredged or remediated.	Floyd Snider McCarthy (2004)
Data report, DSOA vertical characterization and Outfall 12 data collection. Duwamish sediment other area, Boeing Plant 2	DSOAvert char	2001	PCB Aroclors, conventionals[a]	115 samples (impact corer) from 37 locations; 2 to 6 samples at each location; most locations starting at 60 cm to depths of 150 to 280 cm below mudline	Area has not been dredged or remediated.	Pentec (2001)
PSDDA sediment characterization of Duwamish River navigation channel: FY2000 operations and maintenance dredging data report	PSDDA99	1999	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	20 samples (vibracorer) from 20 locations; 3 borings made at each location; 18 samples from 0 to 120 cm; 2 samples from 120 to 240 cm below mudline	Area was proposed for dredging, but no dredging has been conducted since samples were collected.	SEA (2000a, b)
EPA site inspection: Lower Duwamish River	EPA SI	1998	metals, PCB Aroclors and selected congeners, organochlorine pesticides, SVOCs, VOCs, TBT, conventionals[a]	33 samples (vibracorer) from 17 locations; 1 to 2 samples collected at each location, from 0 to 60 and 60 to 120 cm below mudline	None of the areas were dredged after samples were collected, except at RM 0.5 (DR008), which was dredged in 2003/2004.	Weston (1999a)
RCRA facility investigation Duwamish Waterway sediment investigation, Plant 2 – Phase 2b	Plant 2 RFI-2b	1996	metals, PCB Aroclors, SVOCs, conventionals[a]	44 samples (vibracorer) from 16 locations; 2 to 4 samples per core, up to 480 cm below mudline	Area has not been dredged or remediated.	Weston (1998)
RCRA facility investigation Duwamish Waterway sediment investigation, Plant 2 – Phase 1	Plant 2 RFI-1	1995	metals, PCB Aroclors, TPH, SVOCs, VOCs, conventionals[a]	20 samples (vibracorer) from 12 locations; collected at 15- to 45-cm intervals down to 135 cm below mudline	Area has not been dredged or remediated.	Weston (1998)

Note: For reference, 1 ft is equal to 30 cm.

[a] Conventionals may include all or some of the following parameters: grain size, pH, total organic carbon, and total solids.

DSOA – Duwamish sediment other area (Boeing Plant 2)
EAA – early action area
EPA – Environmental Protection Agency
LDW – Lower Duwamish Waterway
O&M – operation and maintenance
PCB – polychlorinated biphenyl
PSDDA – Puget Sound Dredged Disposal Analysis
RCRA – Resource Conservation and Recovery Act
RI – remedial investigation
SVOC – semivolatile organic compound
TBT – tributyltin
TPH – total petroleum hydrocarbons
USACE – US Army Corps of Engineers
VOC – volatile organic compound

Subsurface data collected in areas that have since been dredged, as noted in Table 4-6, were not included in the data summary tables in Section 4.2. These data were excluded because they do not represent existing conditions. However, data from areas that have been dredged are presented on subsurface sediment maps in Section 4.2 to show spatial patterns of chemicals at depth from a historical perspective.

Subsurface samples were collected from different intervals within the cores, depending on the objectives of the different sampling events. Subsurface sediment data are summarized in Section 4.2 according to sampling intervals; these intervals included 1- or 2-ft intervals collected from the uppermost 10 ft of sediment.[44] Eighty-eight percent of the subsurface data was included in the 1- and 2-ft data summaries presented in Section 4.2 (i.e., 786 of the 895 subsurface samples for which data were available from areas that have not been dredged, including calculated data for some of the 2-ft intervals). The remaining 12% of the samples were not included in the 1- and 2-ft interval data summaries because they were not sectioned into 1- or 2-ft intervals. These data are included in the subsurface sediment data summaries, with all intervals combined in Section 4.2.

4.1.2.3 Fish, shellfish, and benthic invertebrate tissue

Tissue chemistry data for the study area are available for multiple tissue types collected as part of several sampling events conducted since 1990. Data from 421 composite samples of fish and shellfish tissue from both sampling events conducted for the RI and historical sampling events were considered suitable for use (Table 4-8). Approximately 74% of these samples were collected in 2004 through 2007 as part of the RI. The remainder of the samples were collected during five other sampling events from 1992 to 2002 and during a King County sampling event in 2006. Data for tissue samples collected in 2006 and 2007 were not included in datasets used in the ERA (Appendix A) or HHRA (Appendix B) because those risk assessments were completed prior to the availability of these data. Collection locations for fish tissue samples are shown on Map 4-9, and collection locations for invertebrate and shellfish samples are shown on Map 4-10.

The RI dataset also included 45 calculated whole-body concentrations: 10 concentrations calculated from English sole fillet and remainder samples collected in 2005 and 35 concentrations calculated from crab edible-meat and hepatopancreas samples collected in 1997, 2004, 2005, and 2007.

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[44] For the calculation of summary statistics, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to the 1- or 2-ft sampling category that best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were also averaged to obtain a calculated concentration for the corresponding 2-ft interval; these intervals were included in addition to the 1-ft intervals.

Table 4-8. Summary of tissue sampling events used in the RI dataset

STUDY	YEAR OF SAMPLING	SPECIES	NO. OF COMPOSITE TISSUE SAMPLES	NO. OF INDIVIDUALS PER SAMPLE	SAMPLE TYPE	USE IN RISK ASSESSMENTS		CHEMICALS
						HHRA	ERA	
LDW RI: chemical analyses of fish, crab, and clam tissue samples collected in 2007 (Windward 2009)	2007	English sole	9	5	skin-on fillet			PCB Aroclors (PCB congeners in a subset of samples), lipids
			19	5	whole body			
		starry flounder	1	5	skin-on fillet			
			3	5	whole body			
		Dungeness crab	4	4 – 5	hepatopancreas[a]			
			4	4 – 5	edible meat			
		slender crab	6	5	hepatopancreas[a]			
			6	5	edible meat			
		shiner surfperch	22	10	whole body			
		eastern softshell clam	15	20 – 23	whole body			total arsenic, inorganic arsenic, lipids (a subset of samples was analyzed for PCB aroclors)
			15	20 – 23	whole body – depurated			
King County fish tissue sampling and chemical analysis in the LDW (Anchor and King County 2007)	2006	shiner surfperch	7	10	whole body			PCB Aroclors, phthalates, lipids
		English sole	6[b]	5				
LDW RI: chemical analyses of fish and crab tissue samples collected in 2005 (Windward 2006c)	2005	English sole	10	5	skin-on fillet	X		PCB Aroclors,[d] lipids
			21	5	whole body[c]	X	X	
		Dungeness crab	3	5	edible meat	X	X	
			3	5	hepatopancreas[a]	X	X	
		slender crab	1	5	edible meat	X	X	
			1	10	hepatopancreas[a]	X	X	
		shiner surfperch	22	10	whole body	X	X	
		Pacific staghorn sculpin	4	10	whole body		X	

Table 4-8, cont. Summary of tissue sampling events used in the RI dataset

Study	Year of Sampling	Species	No. of Composite Tissue Samples	No. of Individuals per Sample	Sample Type	Use in Risk Assessments		Chemicals
						HHRA	ERA	
LDW RI: chemical analyses of fish and crab tissue samples collected in 2004 (Windward 2005b)	2004	English sole	7	5	skin-on fillet	X		metals, SVOCs, organochlorine pesticides, PCB Aroclors (PCB congeners in subset of samples), TBT, lipids
			21	5	whole body	X	X	
		starry flounder	1	5	skin-on fillet	X		
			3	5	whole body	X	X	
		Dungeness crab	7	5	edible meat	X	X	
			3	6 – 15	hepatopancreas[a]	X	X	
		slender crab	12	5	edible meat	X	X	
			4	15 – 18	hepatopancreas[a]	X	X	
		shiner surfperch	24	9 – 10	whole body	X	X	
		striped perch	1	12	skin-on fillet	X		
		pile perch	1	12	skin-on fillet	X		
		Pacific staghorn sculpin	24	7 – 10	whole body		X	
LDW RI: chemical analyses of benthic invertebrate and clam tissue samples and co-located sediment samples (Windward 2005a)	2004	benthic invertebrates	20	nd	whole-body		X	metals, SVOCs, alkylated PAHs, organochlorine pesticides, PCB Aroclors (PCB congeners in subset of samples), TBT, lipids
		eastern softshell clam	14	19 – 52	whole-body	X	X	
LDW RI: juvenile chinook salmon sampling (Windward 2004d)	2004	chinook salmon	18	7 – 12	whole-body		X	metals, SVOCs, alkylated PAHs, organochlorine pesticides, PCB Aroclors, TBT, lipids
			1	72[e]	stomach contents		X	
East Waterway, Harbor Island Superfund site: Tissue chemistry results for juvenile chinook salmon collected from Kellogg Island and East Waterway (Windward 2002)	2002	chinook salmon	6	6-7	whole body		X	mercury, PCB Aroclors, lipids
Waterway Sediment Operable Unit Harbor Island Superfund Site – Assessing human health risks from the consumption of seafood (ESG 1999)	1998	English sole	3	5	skinless fillet	X		mercury, TBT, PCB Aroclors
		red rock crab	2	5	edible meat	X		
		Dungeness crab	1	5	edible meat	X		
		red rock crab/ Dungeness crab	1	5	edible meat	X		
		striped perch	2	5 – 8	skin-on fillet	X		

Table 4-8, cont. Summary of tissue sampling events used in the RI dataset

Study	Year of Sampling	Species	No. of Composite Tissue Samples	No. of Individuals per Sample	Sample Type	Use in Risk Assessments		Chemicals
						HHRA	ERA	
King County Combined Sewer Overflow Water Quality Assessment for the Duwamish River and Elliott Bay (King County 1999e)[f]	1996 – 1997	Dungeness crab	2	3	edible meat	X	X	metals, TBT, SVOCs, PCB Aroclors, lipids
			1	3	hepatopancreas[a]	X	X	
		English sole	3	20	skinless fillet	X		
		shiner surfperch	3	10	whole body		X	
		wild mussels	22	50 – 100	edible meat	X	X	
		transplanted mussels	32	50 – 100	edible meat			
		amphipods	6	nd	whole body		X	
Puget Sound Ambient Monitoring Program – annual sampling (West et al. 2001)	1992	English sole	3	5 – 20	skinless fillet	X		SVOCs, organochlorine pesticides, PCB Aroclors, arsenic, copper, lead, mercury, lipids
	1995	English sole	3	5 – 20	skinless fillet	X		organochlorine pesticides, PCB Aroclors, arsenic, copper, lead, mercury, lipids
	1997	English sole	3	5 – 20	skinless fillet	X		organochlorine pesticides
Elliott Bay/Duwamish River Fish Tissue Investigation (Battelle 1996; Frontier Geosciences 1996)	1995	English sole	3	6	skinless fillet	X		PCB Aroclors, mercury, methylmercury, TBT, lipids

[a] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Total PCB concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data).

[b] Nine individual English sole whole-body tissue samples were also collected and analyzed for PCB Aroclors, phthalates, and lipids.

[c] Whole-body samples include 10 composite samples in which concentrations were calculated using results from separate analyses of fillet tissue and remainder tissue (remainder tissue includes tissue and fluids remaining after fillets were removed from the specimens). The calculated English sole whole-body concentrations were based on the relative weights and total PCB concentrations in skin-on fillet and remainder tissues.

[d] A subset of samples from the 2005 sampling event was also analyzed for PCB congeners. These 2005 PCB congener data are included in the discussion of the nature and extent of PCBs in Section 4.2.3. However, the 2005 data were not included in the risk assessments because the final data were not available at the time those documents were prepared.

[e] An unknown fraction of these stomachs were empty.

[f] Additional samples of cooked crab and English sole were collected during the King County WQA (King County 1999e), but these data were not used.

ERA – ecological risk assessment
HHRA – human health risk assessment
LDW – Lower Duwamish Waterway
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
RI – remedial investigation
TBT – tributyltin

Some additional tissue chemistry data available for the LDW were not included (Table 4-9). Juvenile chinook salmon whole-body and stomach-content data from Varanasi et al. (1993) and from the National Marine Fisheries Service (NMFS) (2002) were available but were not used because the quality assurance/quality control (QA/QC) data were not readily available for EPA review.[45]

Data for adult salmon were excluded because it was estimated that less than 1% of the PCB body burden in adult salmon migrating through the LDW could have been obtained from prey items consumed in the LDW, based on an analysis presented in Appendix B.

Data associated with two individual shiner surfperch collected in Slip 4 by NMFS, 10 individual shiner surfperch collected and analyzed by LDWG in subarea T2E (RM 2.1 to RM 2.4), 10 individual English sole skin-on fillet samples collected and analyzed by LDWG, and 10 individual English sole skinless fillet samples collected and analyzed by LDWG were also excluded (Table 4-9). These data were not included because only data from composite tissue samples were used. The total PCB concentrations in individual fish samples were generally lower than concentrations in composite samples. In addition, juvenile chinook salmon collected upstream of the LDW during the RI were not used; mean PCB concentrations in these fish were lower than those in fish collected in the LDW.[46]

4.1.2.4 Surface water

LDW surface water data are available from three sampling events conducted for the King County water quality assessment (WQA) in 1996 and 1997 (King County 1999e) and during an additional sampling event in 2005, in which water samples were collected and analyzed for PCBs and conventional parameters (Mickelson and Williston 2006) (Table 4-10).

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[45] Total PCB and total DDT concentrations in the NMFS dataset were similar to or lower than concentrations in the RI dataset (Windward 2005f). Total PAH concentrations in LDW juvenile chinook salmon stomach contents from three separate NMFS studies (Arkoosh et al. 1998; McCain et al. 1990; Stein et al. 1995) were higher than those in juvenile chinook salmon stomach contents collected in support of the RI from similar locations in the LDW (Windward 2004d).

[46] Data for upstream sampling were presented in the data report for the juvenile chinook salmon sampling event (Windward 2004d).

Table 4-9. Summary of additional tissue sampling events not included in the RI dataset

STUDY	YEAR OF SAMPLING	SPECIES	SAMPLE TYPE	NUMBER OF INDIVIDUAL OR COMPOSITE SAMPLES	CHEMICALS	REFERENCE
NMFS Duwamish injury assessment project	2000	shiner surfperch	whole body	2[a]	PCBs, pesticides	NMFS (2002)
		juvenile chinook salmon	whole body	29[a, b]		
			stomach contents	6[b]		
Contaminant exposure and associated biochemical effects in outmigrant juvenile chinook salmon from urban and non-urban estuaries of Puget Sound	1989 – 1990	juvenile chinook salmon	whole body	14[b]	PCBs, pesticides, PAHs	Varanasi et al. (1993)
			stomach contents	6[b]		
LDW RI – juvenile chinook salmon sampling	2003	juvenile chinook salmon	whole body	9[b,c]	Metals, PCB Aroclors, PAHs, organochlorine pesticides, lipids	Windward (2004d)
LDW RI – chemical analyses of fish and crab tissue samples	2004	shiner surfperch	whole body	10[d]	PCB Aroclors, lipids	Windward (2004i)
		English Sole	skinless fillet	10[e]	PCB Aroclors, lipids	Windward (2005b)
			skin-on fillet	10[e]	PCB Aroclors, lipids	
Puget Sound Ambient Monitoring Program – annual sampling	1992 – 1998	adult coho and chinook salmon	fillets	153[b]	SVOCs, arsenic, copper, lead, mercury, PCB Aroclors, organochlorine pesticides, lipids	West (2001)

[a] Individual samples were analyzed.

[b] Composite samples were analyzed.

[c] Collected from upstream locations in the Green River at RM 13 and RM 18.

[d] Individual shiner surfperch analyzed from subarea T2E.

[e] Individual English sole fillet samples analyzed from Area T1.

LDW – Lower Duwamish Waterway
PCB – polychlorinated biphenyl
PAH – polycyclic aromatic hydrocarbon
RI – remedial investigation
RM – river mile
SVOC – semivolatile organic compound

Table 4-10. Summary of surface water sampling events included in the RI dataset

SAMPLING EVENT	DATES OF SAMPLING	LOCATION	DEPTHS SAMPLED	SAMPLE TYPE	NUMBER OF SAMPLES	CHEMICALS	REFERENCE
King County WQA	Weekly from October 1996 to June 1997[a]	RM 1.1 (near the Brandon CSO outfall) – east and west banks, mid-channel	1 m below surface and 1 m above river bottom	filtered and unfiltered grab samples	465	antimony, arsenic, cadmium, cobalt, copper, lead, mercury, nickel, selenium, silver, thallium, vanadium, zinc, and SVOCs[c]	King County (1999e)
		RM 1.9 (near the southwest Michigan CSO outfall) – east and west banks, mid-channel					
		RM 4.9 (near the Norfolk CSO outfall) – east and west banks[b]					
King County WQA – SPMD sampling	March 26 to April 8, 1997	Duwamish/Diagonal CSO	1 m and 3 m below surface	SPMDs	4	PCBs, organochlorine pesticides, and PAHs	King County (1999e)
		Brandon Street CSO					
King County 2005	Four dates in 2005: August 22, September 26, November 28, and December 19	RM 0.0 at the south end of Harbor Island	1 m below surface and 1 m above river bottom	unfiltered grab samples	15	209 PCB congeners, total PCBs[d]	King County (2005b), Mickelson and Williston (2006)
		RM 3.3 at the South Park Bridge					

[a] After select storm events, samples were collected on three successive days following the event.

[b] Only two transects were sampled because of the narrow width and shallow water depth in the LDW at this location.

[c] PCBs, pesticides, and PAHs were not analyzed in these surface water samples because of difficulties with obtaining suitably low RLs.

[d] Total PCB concentrations were calculated as the sum of all detected PCB congener concentrations (data were blank-qualified, and RLs for undetected PCB congeners were not included).

CSO – combined sewer overflow
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RM – river mile
SPMD – semipermeable membrane device
SVOC – semivolatile organic compound
WQA – water quality assessment

Grab samples were collected along transects at three locations in the LDW in 1996 and 1997: RM 1.1 (in the vicinity of the Brandon CSO outfall), RM 1.9 (in the vicinity of the Southwest Michigan CSO outfall), and RM 4.9 (in the vicinity of the Norfolk CSO outfall) (Map 4-11a). Samples were collected weekly from October 1996 to June 1997 at all three locations, except during storm events, when sampling was conducted on three successive days following the storm. Chemicals analyzed in both filtered and unfiltered samples included metals and trace elements and semivolatile organic compounds (SVOCs).

PCBs, pesticides, and polycyclic aromatic hydrocarbons (PAHs) were not analyzed in surface water samples collected in 1996 and 1997 because of the difficulty in obtaining suitably low RLs to allow concentrations of these chemicals to be reliably reported.[47] Instead, King County deployed semipermeable membrane devices (SPMDs) at two locations in the LDW from March 26 to April 8, 1997.[48] Three SPMDs, including one field duplicate SPMD, were deployed just offshore of the Duwamish/Diagonal CSO, and two SPMDs were deployed just offshore of the Brandon Street CSO. The SPMDs were analyzed for PCBs, organochlorine pesticides, and PAHs. Chemical concentrations in water were calculated using the mass of chemicals detected in each SPMD and the chemical-specific partition coefficients between the polyethylene and water, which were derived from laboratory experiments conducted at Battelle's Marine Science Laboratory. These calculated concentrations are summarized in Section 4.2 but were not included in the risk assessments because of uncertainty in using calculated concentrations.

In 2005, King County collected water column samples for PCB congener analysis during two dry-weather sampling events in August and September and two wet-weather sampling events in November and December (King County 2005b). Two locations were sampled in the LDW: one location at RM 0.0 at the southern end of Harbor Island, and one at RM 3.3 at the South Park Bridge (Map 4-11a). All 209 PCB congeners were analyzed using high-resolution methods, and total PCB concentrations were calculated as the sum of all detected PCB congener concentrations (data were blank-qualified, and RLs for undetected PCB congeners were not included).

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[47] King County analyzed a limited number of water samples for PCBs, organochlorine pesticides, and PAHs, but all results were reported as not detected. Because obtaining lower detection limits was not possible at that time, analyses for these analytes were discontinued in grab water samples.

[48] The SPMD sampler consisted of layflat, polyethylene tubing, containing a high-molecular-weight neutral lipid. SPMDs mimic the transport across biological membranes and can be used to provide time-averaged chemical concentrations in water, which may include episodic events. Given an adequate deployment time, the concentrations of lipophilic compounds in the SPMD should come close to equilibrium with the concentrations in the water.

4.1.2.5 Seep water

Seep water samples have been collected from shoreline areas along the LDW as part of the RI and other investigations conducted since 1990 (Table 4-11). In addition to the areas sampled for the RI, specific sites that have been sampled include Rhône-Poulenc, Boeing Plant 2, Boeing Isaacson, Terminal 117 (T-117), and GWI (Map 4-11a).

Table 4-11. Summary of seep sampling events included in the RI dataset

SITE	LOCATION	NUMBER OF SEEPS SAMPLED	YEAR OF SAMPLING	SAMPLING METHOD	CHEMICALS	DATA SOURCE
Entire LDW[a]	RM 0.2 to RM 4.3	18	2004	mini-piezometers and funnel with attached tubing to capture direct flow	Metals, SVOCs, VOCs, PCB Aroclors, pesticides, and TPH	Windward (2004b)
T-117[b]	RM 3.5 to RM 3.7 E	3	2003	funnel with attached tubing to capture direct flow	metals, SVOCs, and PCB Aroclors	Windward et al. (2005a)
GWI[c]	RM 2.3 to RM 2.4 E	5	1999	na	SVOCs and VOCs	Terra Vac and Floyd & Snider (2000)
		9	1998	na	VOCs	Terra Vac and Floyd & Snider (2000)
		10	1998	na	VOCs	Terra Vac and Floyd & Snider (2000)
		4	1997	na	VOCs	Terra Vac and Floyd & Snider (2000)
		5	1996	na	VOCs	Terra Vac and Floyd & Snider (2000)
		7	1995	na	VOCs	Terra Vac and Floyd & Snider (2000)
		7	1995	na	VOCs	Terra Vac and Floyd & Snider (2000)
		8	1994	na	VOCs	Terra Vac and Floyd & Snider (2000)
		2	1994	na	VOCs	Terra Vac and Floyd & Snider (2000)
		6	1994	na	VOCs	Terra Vac and Floyd & Snider (2000)
Boeing Plant 2[d]	RM 2.9 to RM 3.6 E	18	1995	capture of direct flow into sampling container	metals, SVOCs, VOCs, and PCB Aroclors, TPH	Weston (1998)

SITE	LOCATION	NUMBER OF SEEPS SAMPLED	YEAR OF SAMPLING	SAMPLING METHOD	CHEMICALS	DATA SOURCE
Rhône-Poulenc[e]	RM 4.0 to RM 4.2 E	7	1995	PVC pipe was placed horizontally out from the bank to capture direct flow; or a small hole was dug and contents were transferred to the sample container.	metals, VOCs, and phthalates	Rhône-Poulenc (1996)
Boeing Isaacson[f]	RM 3.7 E	1	2000	A peristaltic pump was used to pump seep water through a 0.45-micron filter into the sample container.	arsenic	ERM and Exponent (2000)

[a] Both filtered and unfiltered samples were analyzed for metals, PCBs, SVOCs, and pesticides; samples analyzed for VOCs and TPH were unfiltered.

[b] Both filtered and unfiltered samples were analyzed for metals; samples analyzed for PCBs and SVOCs were unfiltered. One seep was re-sampled in 2004 for PCB analysis; this sample was centrifuged.

[c] A total of 10 seeps were sampled; some of the seeps were sampled on more than one date; it is not known if samples were filtered or unfiltered.

[d] Both filtered and unfiltered samples were analyzed for metals; samples analyzed for PCBs and SVOCs were unfiltered.

[e] Data are available only for detected chemicals (metals, BEHP, and formaldehyde); samples were unfiltered.

[f] Samples were filtered.

BEHP – bis(2-ethylhexyl) phthalate
GWI – Great Western International
LDW – Lower Duwamish Waterway
na – not available
PCB – polychlorinated biphenyl
PVC – polyvinyl chloride
RM – river mile
SVOC – semivolatile organic compound
T-117 – Terminal 117
TPH – total petroleum hydrocarbon
VOC – volatile organic compound

4.1.2.6 Porewater

Table 4-12 summarizes the available porewater datasets collected from the LDW. Samples were collected from GWI and Boeing Plant 2/Jorgensen Forge in 2005 and analyzed for volatile organic compounds (VOCs) as part of the RI. At Rhône-Poulenc, porewater samples were collected in 2004 and were analyzed for metals and organic compounds. As part of the EPA site inspection (SI), porewater samples were collected in 1998 throughout the LDW and were analyzed for metals. Porewater samples were collected and analyzed for butyltins during five sampling events: Glacier Northwest, Duwamish Yacht Club, Hurlen-Boyer, PSDDA99, and PSDDA98. Data from the PSDDA99 sampling event are discussed in this RI; samples from the other four events are not discussed because they were collected from areas that have been dredged and thus do not represent current conditions. Porewater locations sampled during the GWI, Boeing Plant 2/Jorgensen Forge, Rhône-Poulenc, EPA SI, and PSDDA99 sampling events are shown on Map 4-12a.

Table 4-12. Summary of porewater sampling events

SITE OR SAMPLING EVENT	LOCATION	NUMBER OF SAMPLES	YEAR OF SAMPLING	SAMPLING METHOD	CHEMICALS	REFERENCE
Great Western International	RM 2.3 to RM 2.4 E	14	2005	mini-piezometer and peeper	VOCs	Windward (2006a)
Boeing Plant 2/ Jorgensen Forge	RM 3.6 E	14	2005	mini-piezometer and peeper	VOCs	Windward (2006a)
Rhône-Poulenc	RM 4.0 to RM 4.2 E	16	2004	mini-peizometer and seepage meter	metals, BTEX, PAHs, phthalates	EPA (2005d)
EPA SI	entire LDW	15	1998	centrifuge from sediment	metals and butyltins	Weston (1999a)
Glacier Northwest[a]	RM 1.4 to RM 1.5 W	5	2002	na	butyltins	PIE (2002)
Duwamish Yacht Club[a]	RM 4.1 W	2	1999	na	TBT	Hart Crowser (1999)
PSDDA99	RM 1.9 to RM 3.4 W	20	1999	na	TBT	SEA (2000b)
Hurlen-Boyer[a]	RM 2.4 to RM 2.7 W	2	1998	centrifuge from sediment	TBT	Hart Crowser (1998)
PSDDA98[a]	RM 4.4 to RM 4.6 in the navigation channel	2	1998	centrifuge from sediment	TBT	SEA (1998)

[a] Data from these events are not summarized in this document because sediment was dredged after porewater samples were collected.

BTEX – benzene, toluene, ethylbenzene, and xylene
EPA – US Environmental Protection Agency
LDW – Lower Duwamish Waterway
na – not available
PAH – polycyclic aromatic hydrocarbon
PIE – Pacific International Engineering, PLLC
PSDDA – Puget Sound Dredged Disposal Analysis
RM – river mile
SEA – Striplin Environmental Associates, Inc.
SI – site inspection
TBT – tributyltin
VOC – volatile organic carbon

4.1.3 Data suitability

Once data were determined to meet met DQOs, the suitability of data for specific purposes in the RI was evaluated according to EPA guidance (EPA 1988, 1992b). Of primary importance was the degree to which the data adequately represented site-related contamination and potential exposure of humans and ecological receptors at the site. Other considerations in determining the suitability of datasets that met DQOs included the availability of data validation results, whether results may have been confounded by analytical interferences, and whether data represented a significant departure from the standard analytical methods used for other samples in the dataset.

4.1.3.1 Representativeness

This section provides a discussion of whether the available sediment, tissue, and water data represent the ranges and distributions of chemical concentrations in the LDW.

Surface Sediment

Many historical environmental sampling events have included the collection and analysis of surface sediment from the LDW (Table 4-4). These studies were designed for site-wide or reconnaissance surveys (e.g., Boeing SiteChar, EPA SI, and NOAA SiteChar) or focused investigations of suspected areas of contamination (e.g., Boeing Resource Conservation and Recovery Act [RCRA] facility investigation [RFI], Rhône-Poulenc RFI). Because many of the sediment samples were collected to characterize contaminated areas, the dataset as a whole may have a relatively higher proportion of elevated chemical concentrations than is representative of site-wide conditions within the LDW. Of the 1,365 sampling locations in the RI baseline surface sediment dataset, 880 of these locations were sampled as part of a site-wide sampling event, and 485 locations were sampled during focused investigations, as shown on Maps 4-1 and 4-2, respectively.

Subsurface Sediment

Many historical environmental sampling events have included the collection of subsurface sediment from the LDW (Table 4-6). Of the 895 subsurface samples (from 309 cores that have not been dredged), 265 samples (from 78 cores) were collected as part of site-wide events for a more general characterization of the nature and extent of contamination. Of these 265 samples, 234 samples (from 62 cores) were collected in 2006 as part of the RI and 31 samples (from 16 cores) were collected during the EPA SI in 1998 (Map 4-6). The remaining 630 samples (from 231 cores) were collected from areas of suspected contamination (Map 4-5). Because many of the subsurface sediment samples were collected to characterize contaminated areas, the dataset as a whole likely has a higher proportion of samples with chemical concentrations that are elevated relative to the mean chemical concentrations expected throughout the entire LDW.

Tissue

To represent chemical concentrations in tissue within the LDW, data for various biological tissue types are necessary (Table 4-8). In addition, a representative set of tissue data should include spatial coverage for each species according to its home range and appropriate size classes to meet specific objectives of sampling for each organism, and should contain a sufficient number of samples to represent differing exposure regimes within the LDW.

To supplement tissue data for the LDW that were collected prior to the RI, large numbers of fish and crabs were collected as part of the RI data collection efforts for use primarily in the risk assessments (Windward 2004e, f).[49] These data were also used to summarize the nature and extent of chemical concentrations in biological matrices of the LDW. Species were selected to represent exposure of humans and ecological receptors, and thus various fish and crab species were collected. Samples collected for wildlife exposure included the entire specimen, while samples collected for human exposure included only the edible portions of fish or crab species (i.e., fillet and edible meat). The tissue sampling events conducted in 2004, 2005, 2006, and 2007 provided 28 to 74 composite samples for each of three fish species for whole-body tissue, 39 composite samples of English sole or starry flounder fillet tissue, 33 composite samples of crab edible meat tissue, and 21 composite samples of crab hepatopancreas tissue, all of which satisfied the minimum number of samples (i.e., six) that EPA, Ecology, and LDWG agreed was acceptable for calculation of a UCL for each chemical and tissue type for the LDW (Windward 2004f).

Tissue sampling performed for the RI was designed to generate data representative of the range of concentrations of key chemicals to which an organism may be exposed and to provide spatial coverage of the LDW. For mobile species such as fish and crabs, spatial representation of the LDW was accomplished by collecting samples from four sampling areas distributed throughout the LDW. These areas were selected to cover a range of rolling average concentrations of total PCBs in sediment, as discussed in detail in the fish and crab QAPP (Windward 2004f). For benthic invertebrates, 20 sampling locations were chosen to represent areas with a range of concentrations of total PCBs, arsenic, and lead in sediment, as well as to provide spatial coverage of intertidal areas in the LDW. For clams, 10 sampling locations were placed in known clam habitat areas to cover the concentration range of PCBs, cPAHs, and arsenic in sediment (Windward 2004e). Target sizes of fish and crabs collected during the RI were selected to represent the prey size ranges preferred by piscivorous wildlife receptors and reasonable size ranges of seafood consumed by people (Windward 2004f).

Surface Water

Surface water samples collected during the King County WQA and analyzed for metals and SVOCs were part of a large set of water samples collected weekly from October 1996 to June 1997 (and for three successive days following select storm events) along transects at three sites in the LDW (see Section 4.1.2.4) (King County 1999e). These data represent a seasonal range of flow conditions in the LDW. Time-integrated samples were also collected at two locations and two depths for PCBs, pesticides, and

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[49] The 2006 and 2007 data were collected to evaluate changes in PCB concentrations over time; these data were not used in the risk assessments because they were finalized after the HHRA and ERA were completed.

PAHs using SPMDs during the King County WQA. In addition, PCB congeners were analyzed by King County in whole-water samples collected in 2005 during two dry-weather periods (August and September) and two wet-weather periods (November and December) at two locations and two depths. The PCB congener data represent a limited number of specific areas. They were assumed to represent general conditions within these two areas of the LDW because the LDW is a well-mixed tidal estuary; however, these data may not represent the full range of total PCB concentrations in water for all locations or seasons because of localized conditions or short-term fluctuations arising from a myriad of factors.

SPMDs, which are pre-cleaned polyethylene sheets, were deployed in the LDW over a period of 13 days. During deployment, PCBs, pesticides, and PAHs in the water column were sorbed onto the SPMDs. The SPMDs were extracted and analyzed using standard test methods, and compound-specific partitioning coefficients, if available, were applied to the resultant SPMD data to estimate the average surface water concentrations. PCBs were analyzed as both PCB congeners (17 of the 209 PCB congeners) and Aroclors. The total PCB concentrations calculated as PCB congener sums likely underestimate total PCBs because only a subset of the 209 congeners were analyzed, so these data were not used in the RI or the ERA. Instead, water samples collected by King County in 2005 that were analyzed for all 209 congeners were used (Mickelson and Williston (2006). The Aroclor data from SPMDs were not used in the RI or the ERA because non-Aroclor-specific partitioning coefficients were used, so the calculated concentrations are highly uncertain. There are also uncertainties associated with pesticide and PAH water concentrations back-calculated from the SPMD data. However, they are presented in the RI for informational purposes because no other surface water data are available for those compounds. Surface water data were not used to characterize risk from pesticides or PAHs in the ERA.

The water data were used in the ERA to estimate the exposure of fish and wildlife to chemicals through water ingestion. The surface water data were considered sufficient for use in the ERA because: 1) mean concentrations were of interest for the ERA because they represent chronic exposures ; 2) the exposure of fish to PCBs and pesticides was evaluated through a critical tissue-residue approach, which incorporated exposure through the water column, and 3) the ingestion of chemicals via water is a very minor component of the total ingestion of chemicals by birds and mammals from prey, sediment, and water combined (Appendix A, Section A.5.1.3.3).

The water data were also used to assess the nature and extent of contamination in the LDW and for use in the food web model (FWM). The PCB congener data collected in 2005 were also used to update King County's EFDC model, a hydrodynamic and chemical fate and transport model (Mickelson and Williston 2006). The EFDC model estimates PCB concentrations in water for the entire LDW over different spatial scales and over various temporal scales that include all four seasons. The results of the updated EFDC model were used in the RI to refine the predictions of total PCB concentrations in the LDW water column over monthly time periods at two different spatial scales for the FWM (Appendix D). In addition, total PCB concentrations calculated as the sum of PCB congeners from these datasets were used to define the upper and lower limits of total PCB concentrations in water for the FWM. These upper and lower limits were used to represent LDW-wide mean PCB concentrations in the water. The FWM was used to estimate sediment risk-based threshold concentrations (RBTCs) for total PCBs. Appendix D addresses the sufficiency of the surface water data for updating the EFDC model and for use in the FWM.

Porewater

Porewater samples were collected as part of the RI to evaluate risks to the benthic invertebrate community [50] in areas assumed to represent worst-case exposure areas based on potential exposure to groundwater contamination. Porewater data were collected from two areas of the LDW (GWI and Boeing Plant 2/Jorgensen Forge) to investigate potential worst-case conditions for discharge of VOCs from the groundwater (Windward 2006a). Other areas in the LDW were assumed to have a lower potential for groundwater discharge of VOCs based on existing source data. Detailed groundwater conceptual site models were developed for both the GWI and Boeing Plant 2/Jorgensen Forge sites to ensure that sampling was conducted in areas with the highest potential for groundwater discharge (Windward 2005h). Porewater samples were collected using peepers deployed 10 cm beneath the sediment surface and left to equilibrate for 2 weeks. Because of their placement within the biologically active zone and the equilibration time allowed, these samples should reasonably represent benthic invertebrate exposure conditions within these areas.

Other historical porewater data provide information about potential sources in the LDW and were therefore included in the characterization of nature and extent of contamination. The Rhône-Poulenc investigation (EPA 2005d) collected porewater data for metals and SVOCs from locations along the shoreline of the facility using seepage meters and piezometers. The locations and methods for sampling porewater were designed to represent groundwater or surface water in the subsurface-surface transition zone from areas most likely to be contaminated from the Rhône-Poulenc site.

The EPA SI collected porewater data for metals from 15 locations throughout the LDW by centrifuging the porewater from sediment as part of a screening-level investigation of the LDW (Weston 1999a). The objectives for selecting these locations were not presented in the EPA SI documents; it is assumed that these samples were collected to represent a range of conditions throughout the LDW. Although porewater samples

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[50] Because VOCs have a low affinity for sediment, analysis of porewater is the best way to estimate exposure. The primary use of porewater data in the RI was for the evaluation of risk to benthic invertebrates from exposure to VOCs. For other chemicals that have a higher affinity for sediment, risk to benthic invertebrates was better addressed using sediment chemistry data.

were analyzed for butyltins during five sampling events, data from four of these events were collected from areas that were subsequently dredged, so they do not represent current conditions. In the remaining sampling event, 20 samples were collected between RM 1.9 to RM 3.4. Note that different methods of porewater collection may generate different results.

Seep Water

As part of the RI, a reconnaissance survey was conducted in May 2004 to identify seeps along the entire LDW shoreline (Windward 2004b). Eighteen seeps from throughout the LDW were selected for collection and chemical analysis of seep water samples. Seeps were selected based on an evaluation of the proximity of the sampling locations to potential sources, surface sediment chemistry, and observations made during the reconnaissance survey. The primary objective of this sampling was to characterize suspected sources of contaminants to the LDW. Although rainfall was lower than usual in the spring of 2004 an analysis of expected groundwater elevations indicated that the lower rainfall would not have affected the ability to identify or sample seeps at that time (Windward 2004g)

In addition to the 18 seep locations sampled as part of the RI, seep data were available for four other sites along the LDW. Available data were used to describe the nature and extent of contamination in seep water.

4.1.3.2 QA/QC results

All datasets used in the RI (i.e., datasets for sediment, tissue, surface water, porewater, and seep water) were validated by the original authors of the individual studies or by third parties. Summaries of these data validations for historical sampling events have been presented in technical memoranda prepared by Windward (Windward 2005k, l, 2007e). Data validation reports for samples collected as part of the RI were included in the data reports for each sampling event. The only surface sediment chemistry dataset not considered acceptable for all uses in the RI/FS because the DQOs were not met were data collected from the Rhône-Poulenc facility in 1994; these data were not considered acceptable because Form 1s were not available.

Based on data quality, approximately 1,000 analytical results, across all media types, were qualified as unusable out of more than 150,000 analytical results. These data were excluded because of concerns such as extremely low matrix or surrogate spike recoveries, identified by the analytical laboratory or by the data validator. Almost all of the rejected results were for SVOCs (i.e., 1,2,4-trichlorobenzene, 1,2-dichlorobenzene, 2,4-dimethylphenol, 3,3′-dichlorobenzidine, 3-nitroaniline, 4-chloroaniline, 4-nitroaniline, aniline, benzidine, benzoic acid, hexachlorocyclopentadiene, pentachlorophenol, and pyridine). In addition, some sediment results for metals (i.e., antimony, cadmium, silver, and selenium) were rejected, as were several results for organochlorine pesticides (i.e., delta-benzene hexachloride [BHC], endosulfan sulfate, and endrin aldehyde). Data qualified as unusable were not used in the risk assessments or presented in the RI.

Analytical interference with the quantification of organochlorine pesticides from the presence of PCB congeners occurred during the analyses of benthic invertebrate tissue (Windward 2005a) and fish and crab tissue (Windward 2005b). This issue was identified by both the analytical laboratory and the data validator. The organochlorine pesticides were analyzed using EPA Method 8081A (gas chromatography with electron capture detection [GC/ECD]), which is a standard method used in many environmental investigations for organochlorine pesticides. The detected results for both the benthic invertebrate and clam tissue samples collected in 2004 (Windward 2005a) and the fish and crab tissue samples collected in 2004 (Windward 2005b) were qualified JN, which indicates "the presence of an analyte that has been 'tentatively identified' and the associated numerical value represents its approximate concentration" (EPA 1999c). These data were qualified based on the probable interference in the organochlorine pesticide analysis from PCB congeners.

The JN-qualified results are highly uncertain and biased high. The high bias for DDTs (dichlorodiphenyltrichloroethane) was confirmed by re-analyzing six sediment samples co-located with benthic invertebrate tissue samples and eight fish and crab tissue samples that had high PCB and DDT concentrations using a gas chromatography/mass spectrometry (GC/MS) method, which is not susceptible to analytical interference by PCBs for organochlorine pesticides. Additional details regarding the confirmation analysis results are presented in Section A.2.4.2 of Appendix A. Both the HHRA and the ERA address the uncertainty associated with the use of the JN-qualified organochlorine pesticide data in evaluating risks to human and ecological receptors. This uncertainty is also relevant to the nature and extent characterization of organochlorine pesticides in sediment.

4.1.3.3 Analytical Methods

The sediment surveys from which the RI baseline surface sediment chemistry dataset was compiled used similar or identical analytical methods for most analytes, with one notable exception. PCB analyses for the NOAA site characterization were conducted using high-performance liquid chromatography/photodiode array detection (HPLC/PDA). This method was not used for PCB Aroclor analyses in any of the other events; other analyses were conducted using GC/ECD. NOAA laboratory data for total PCBs are based on a nonstandard analytical method and may not be quantitatively comparable to data generated using standard analytical techniques. Specifically, the NOAA laboratory data for total PCBs reflect the difference between the sum of PCBs and polychlorinated terphenyls (PCTs) and the results of a separate analysis for PCTs alone.

Krahn et al. (1998) reported the results for 30 samples that were analyzed using both HPLC/PDA and GC/ECD methods by two different laboratories.[51] The two laboratories calculated total PCBs for each sample, which were then compared with each other. Total PCB concentrations varied between the two laboratories by as much as a factor of 6 (Krahn et al. 1998). Regression analyses conducted for the two sets of results indicated that the GC/ECD results were lower than the HPLC/PDA results at high PCB concentrations and higher than the HPLC/PDA results at low PCB concentrations (Krahn et al. 1998). The regression coefficient (R^2) between the two sets of analyses was 0.92. The differences between the total PCB concentrations calculated by the two laboratories are not surprising given the differences between the two methods, including: 1) different ranges of linear response for the two detectors, 2) differences in methods for calculating total PCBs, 3) differences in methods for quantifying and/or removing analytical interferences, and 4) differences in RLs.

Despite the differences between the two analytical methods for PCBs, data from both methods were used in the RI and risk assessments, although the uncertainty associated with total PCB concentrations may be significant in some areas. Alternative risk calculations are presented in the HHRA (Appendix B, Section B.6) using total PCB data derived solely from summing PCB Aroclor concentrations analyzed using GC/ECD.

Although different laboratories, and in some cases different methods, were used for the various tissue analyses, all tissue data summarized in Table 4-8 met the DQOs established for the project (Section 4.1). The uncertainties associated with combining data from different events that used different analytical methods are described in more detail in the HHRA (Appendix B, Section B.6).

4.1.4 Data reduction

Data reduction refers to methods used to aggregate raw data received from the laboratory for use in the RI/FS. A detailed discussion of data reduction methods is presented in Appendix E, Section E.3, and briefly summarized as follows:

- Chemical concentrations obtained from the analysis of laboratory duplicates or replicates (i.e., two or more analyses of the same sample) were averaged.
- Chemical concentrations of discrete samples collected at a single sampling location that were submitted to the laboratory as individual samples and analyzed separately were averaged and evaluated as a single sample.
- In some instances, the laboratory generated more than one result for a chemical for a given sample if re-analysis was required or if two different analytical

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[51] HPLC/PDA analyses were conducted by the NOAA laboratory in Seattle; GC/ECD analyses were conducted by ARI in Seattle.

methods were used for that chemical. The procedures for selecting the best result are described in Appendix E, Section E.3.

- The precision of each result was stored in the project database by recording the number of significant figures assigned by the laboratory. These significant figures were treated according to methods described in Appendix E, Section E.3.

For several chemicals (PCBs, DDTs, PAHs, and chlordane), total concentrations were calculated in individual samples by summing concentrations of individual components (i.e., seven Aroclor mixtures or individual congeners for total PCBs, six DDT isomers for total DDTs, specific individual PAH compounds for total PAHs, and specific individual chlordane compounds for total chlordane). The treatment of non-detects for these sums were as follows:

- If some of the individual components were detected in a sample and some were not, only the detected concentrations were included in the sum.
- If none of the individual components were detected in a sample, the total concentration was given a value equal to the highest RL of an individual component and assigned a U-qualifier, indicating the lack of detected concentrations.

When calculating the mean concentration for a group of samples for chemicals expressed as totals (e.g., Aroclors, DDT isomers, individual PAH compounds, or individual chlordane compounds), if none of the individual components of the mixture were detected in a given sample, then one-half the reported non-detect value was used for that sample. Calculation of the reported non-detect value is discussed in Appendix E. Toxic equivalents (TEQs) of dioxins and furans and PCBs were calculated by summing the products of concentrations and compound-specific toxic equivalency factors (TEFs) for individual congeners of polychlorinated dibenzo-*p*-dioxins and furans or PCBs, as discussed in more detail in Appendix E, Section E.3. Total cPAHs were calculated by summing the products of concentrations and compound-specific potency equivalency factors (PEFs) for individual cPAH compounds, also discussed in more detail in Appendix E, Section E.3. Congeners or individual cPAH compounds that were undetected for a given sample were assigned a value equal to one-half the sample-specific RL for use in the calculations. Only the congeners or compounds that were analyzed were included in the TEQ or cPAH calculations.

4.2 CHEMICAL-SPECIFIC NATURE AND EXTENT INFORMATION

This section presents the nature and extent of contamination in surface sediment, subsurface sediment, tissue, surface water, seep water, and porewater, including maps that show distributions of selected chemicals in the various media. To focus the presentation of sediment data, a chemical was included if it exceeded the SQS of the

SMS or was identified as a chemical of concern (COC) in the HHRA (Appendix B)[52] based on a sediment exposure pathway (as described in Sections 4.2.1.2 and 6.5, respectively). In tissue, a chemical was included if it was a COC for any fish or wildlife receptor in the ERA (Appendix A) or if it was identified as a COC in the HHRA based on seafood consumption (as described in Sections 5.5 and 6.5, respectively). In addition, data summary tables for tissue included any detected SVOCs and all organochlorine pesticides. All tissue data collected between 1992 and 2007 are included in the data summary tables. In general, data summary tables for surface water, seep water, and porewater included any chemicals with detected concentrations. Data for all chemicals in each media are included in Appendix E.

This section begins with a general LDW-wide comparison of all sediment chemistry data to the SQS and CSL criteria of the SMS. This SMS discussion is followed by chemical-specific descriptions of the nature and extent of contamination in various site media, organized as follows: Sections 4.2.3 through 4.2.7 present discussions of five risk driver chemicals (PCBs, arsenic, cPAHs, BEHP, and dioxins and furans) and Sections 4.2.8 through 4.2.11 present discussions for metals and tributyltin (TBT), SVOCs, organochlorine pesticides, and VOCs, respectively.

Summaries presented in this section are based on data of acceptable quality collected from 1990 to the present, as described in Section 4.1. Summary statistics for all chemicals analyzed in each media are presented in Appendix E, Section E.6. Percentiles of five risk driver chemical concentrations in the RI baseline surface sediment dataset were calculated to facilitate data presentation. These percentiles represent the concentration below which the corresponding percent of individual concentrations in the dataset fall (e.g., the 95th percentile is the concentration below which 95% of the concentrations fall). These percentiles are referred to as numerical percentiles. For some chemicals (i.e., PCBs, arsenic, cPAHs, and BEHP), area-based percentiles were also calculated based on either inverse distance weighting (IDW) or Thiessen polygon interpolations to illustrate the spatial distribution of these chemicals. Area-based percentiles are calculated in the same way as numerical percentiles, except the percentiles are established by the percentage of the total LDW area, rather than the range of concentrations (e.g., the 95th percentile is the concentration at which 95% of the LDW area has interpolated values below that concentration). Both numerical and area-based percentiles were based on dry weight (rather than OC-normalized) concentrations.

Area-based percentiles are discussed in the context of maps that present the interpolated concentrations. Where only point-based maps are presented, numerical percentiles are discussed.

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[52] A chemical was identified as a COC in the HHRA if the excess cancer risk estimate was greater than 1×10^{-6} or if the non-cancer HQ was greater than 1 for at least one reasonable maximum exposure scenario.

Appendix C presents the UCLs for arsenic, total PCBs, cPAHs, and dioxins and furans using the RI baseline surface sediment dataset and the same calculation methods as those used in the HHRA (Appendix B).[53] The FS presents another set of UCLs calculated using data from the FS dataset and a spatially weighted method.

4.2.1 Sediment chemistry and toxicity relative to SMS criteria

This section describes the SMS marine sediment criteria (WAC 173-204) followed by comparisons of surface and subsurface chemistry data and toxicity test results with those criteria. The discussion in this section presents the locations with exceedances of SMS criteria (for any chemical); more detailed information on the locations and extent of SQS/CSL exceedances for specific chemicals or chemical groups is provided in the chemical-specific sections that describe nature and extent of contamination (Sections 4.2.3 through 4.2.9).

4.2.1.1 SMS criteria

The SMS regulations provide both chemical and biological effects-based criteria. The numerical SMS chemical criteria are available for 47 chemicals or groups of chemicals. The SQS represents a numerical chemical concentration below which sediments are designated as having no adverse effects on biological resources. At chemical concentrations above the SQS but below the CSL, sediments are designated as having minor adverse effects on biological resources. At chemical concentrations above the CSL, there is a potential for more pronounced adverse affects. The SMS chemical criteria for the 47 chemicals (or groups of chemicals) are presented in Table 4-13.

Table 4-13. SMS chemical criteria for marine sediments

CHEMICAL	UNIT	SQS	CSL
Metals			
Arsenic	mg/kg dw	57	93
Cadmium	mg/kg dw	5.1	6.7
Chromium	mg/kg dw	260	270
Copper	mg/kg dw	390	390
Lead	mg/kg dw	450	530
Mercury	mg/kg dw	0.41	0.59
Silver	mg/kg dw	6.1	6.1
Zinc	mg/kg dw	410	960
PAHs			
2-Methylnaphthalene	mg/kg OC	38	64
Acenaphthene	mg/kg OC	16	57
Acenaphthylene	mg/kg OC	66	66

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[53] These UCLs are presented as exposure point concentrations for the netfishing scenario in the HHRA (Appendix B), which uses the entire surface sediment dataset, in Table C.3-8 of Appendix C.

CHEMICAL	UNIT	SQS	CSL
Anthracene	mg/kg OC	220	1,200
Benzo(a)anthracene	mg/kg OC	110	270
Benzo(a)pyrene	mg/kg OC	99	210
Benzo(g,h,i)perylene	mg/kg OC	31	78
Total benzofluoranthenes	mg/kg OC	230	450
Chrysene	mg/kg OC	110	460
Dibenzo(a,h)anthracene	mg/kg OC	12	33
Dibenzofuran	mg/kg OC	15	58
Fluoranthene	mg/kg OC	160	1,200
Fluorene	mg/kg OC	23	79
Indeno(1,2,3-cd)pyrene	mg/kg OC	34	88
Naphthalene	mg/kg OC	99	170
Phenanthrene	mg/kg OC	100	480
Pyrene	mg/kg OC	1,000	1,400
Total HPAH[a]	mg/kg OC	960	5,300
Total LPAH[b]	mg/kg OC	370	780
Phthalates			
BEHP	mg/kg OC	47	78
BBP	mg/kg OC	4.9	64
Diethyl phthalate	mg/kg OC	61	110
Dimethyl phthalate	mg/kg OC	53	53
Di-n-butyl phthalate	mg/kg OC	220	1,700
Di-n-octyl phthalate	mg/kg OC	58	4,500
Other SVOCs			
1,2,4-Trichlorobenzene	mg/kg OC	0.81	1.8
1,2-Dichlorobenzene	mg/kg OC	2.3	2.3
1,4-Dichlorobenzene	mg/kg OC	3.1	9.0
2,4-Dimethylphenol	µg/kg dw	29	29
2-Methylphenol	µg/kg dw	63	63
4-Methylphenol	µg/kg dw	670	670
Benzoic acid	µg/kg dw	650	650
Benzyl alcohol	µg/kg dw	57	73
Hexachlorobenzene	mg/kg OC	0.38	2.3
Hexachlorobutadiene	mg/kg OC	3.9	6.2
n-Nitrosodiphenylamine	mg/kg OC	11	11
Pentachlorophenol	µg/kg dw	360	690
Phenol	µg/kg dw	420	1,200
PCBs			
Total PCBs	mg/kg OC	12	65

[a] Total HPAHs were calculated as the sum of benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, total benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, and pyrene.

[b] Total LPAHs were calculated as the sum of acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
CSL – cleanup screening level
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
OC –organic carbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SMS – Washington State Sediment Management Standards
SQS – sediment quality standards
SVOC – semivolatile organic compound

Many of the SQS and CSL criteria are in units normalized to the organic carbon content in the sediment sample (mg/kg OC). Concentrations originally in units of µg/kg dry weight were converted to mg/kg OC using the following equation:

$$C_{OC} = \frac{C_{dw}}{TOC} \qquad \textbf{Equation 4-1}$$

Where:

C_{dw} = dry weight chemical concentration (mg/kg dw)
C_{OC} = OC-normalized chemical concentration (mg/kg OC)
TOC = fraction of total organic carbon

OC normalization is not considered to be appropriate for TOC concentrations ≤ 0.5% or ≥ 4.0% (Michelsen and Bragdon-Cook 1993). In these cases, dry weight chemical concentrations were compared with the lowest apparent effects threshold (LAET), which is functionally equivalent to the SQS, or the second lowest AET (2LAET), which is functionally equivalent to the CSL. The LAET and 2LAET values for chemicals with SMS criteria that are OC normalized are presented in Table 4-14.

Table 4-14. LAETs for chemicals with OC-normalized SMS criteria

CHEMICAL	CONCENTRATION (µg/kg dw)	
	LAET	2LAET
PAHs		
2-Methylnaphthalene	670	1,400
Acenaphthene	500	730
Acenaphthylene	1,300	1,300
Anthracene	960	4,400
Benzo(a)anthracene	1,300	1,600
Benzo(a)pyrene	1,600	3,000
Benzo(g,h,i)perylene	670	720
Total benzofluoranthenes	3,200	3,600
Chrysene	1,400	2,800
Dibenzo(a,h)anthracene	230	540
Dibenzofuran	540	700
Fluoranthene	1,700	2,500

Chemical	Concentration (µg/kg dw)	
	LAET	2LAET
Fluorene	540	1,000
Indeno(1,2,3-cd)pyrene	600	690
Naphthalene	2,100	2,400
Phenanthrene	1,500	5,400
Pyrene	2,600	3,300
Total HPAH[a]	12,000	17,000
Total LPAH[b]	5,200	13,000
Phthalates		
BEHP	1,300	1,900
BBP	63	900
Diethyl phthalate	200	1,200
Dimethyl phthalate	71	160
Di-n-butyl phthalate	1,400	5,100
Di-n-octyl phthalate	6,200	nv
Other SVOCs		
1,2,4-Trichlorobenzene	31	51
1,2-Dichlorobenzene	35	50
1,4-Dichlorobenzene	110	120
Hexachlorobenzene	22	70
Hexachlorobutadiene	11	120
n-Nitrosodiphenylamine	28	40
PCBs		
Total PCBs	130	1,000

[a] Total HPAHs were calculated as the sum of benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, total benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, and pyrene.

[b] Total LPAHs were calculated as the sum of acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LAET – lowest apparent effects threshold
2LAET – second lowest apparent effects threshold
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
nv – no value
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SMS – Washington State Sediment Management Standards
SVOC – semivolatile organic compound

The SMS regulations also include biological criteria (Washington State Administrative Code [WAC] 173-204) based on sediment toxicity tests or benthic infaunal abundance. Because AETs, which form the basis for the chemical criteria, are based on sediment samples with a mixture of chemicals from various locations in Puget Sound and exceedance of the SMS chemical criteria is not always an accurate predictor of adverse effects, the regulations state that site-specific biological tests (sediment toxicity tests or assessment of benthic infaunal abundances) may be conducted to provide confirmation that site-specific chemistry data indicate a hazard to benthic invertebrate communities. The SQS and CSL biological effects criteria for the toxicity tests conducted for the RI are presented in Table 4-15; there were no assessments of benthic infaunal abundances, so effects criteria for those assessments are not included. The SQS is exceeded if the SQS biological criteria are exceeded for any one of the toxicity tests. Likewise, the CSL is exceeded if the CSL biological criteria are exceeded for any one of the toxicity tests. The CSL is also exceeded if the SQS biological effects criteria are exceeded in any two toxicity tests at one location (WAC 173-204-420(3)). The SQS and CSL designations based on biological criteria override the SQS and CSL designations based on chemistry results. For example, if a sample has a chemical SQS exceedance but is tested and found not to be toxic, it is not categorized as an SQS exceedance.

Table 4-15. SMS biological effects criteria for marine sediment toxicity tests conducted for the RI

TOXICITY TEST	BIOLOGICAL EFFECTS CRITERIA	
	SQS	CSL
Amphipod	mean mortality > 25% on an absolute basis and statistically different from the reference sediment ($p \leq 0.05$)	mean mortality greater than the value in the reference sediment plus 30% and statistically different from the reference sediment ($p \leq 0.05$)
Polychaete[a]	mean individual growth rate < 70% of that of the reference sediment and statistically different ($p \leq 0.05$)	mean individual growth rate < 50% of that of the reference sediment and statistically different ($p \leq 0.05$)
Bivalve larvae	mean normal survivorship < 85% of that of the reference sediment and statistically different ($p \leq 0.10$)	mean normal survivorship < 70% of that of the reference sediment and statistically different ($p \leq 0.10$)

[a] The mortality endpoint for the polychaete toxicity test is not used for determination of SMS compliance.

CSL – cleanup screening level

SMS – Washington State Sediment Management Standards

SQS – sediment quality standards

4.2.1.2 Surface sediment chemistry relative to SMS

Table 4-16 presents a summary of SMS criteria and SQS and CSL exceedances for surface sediment samples in the RI baseline dataset. Detected concentrations of 39 SMS chemicals exceeded the CSL at one or more locations (Map 4-13). Two additional chemicals (i.e., anthracene and pentachlorophenol) were detected at concentrations that exceeded the SQS but did not exceed the CSL. Detected concentrations of the remaining six SMS chemicals (i.e., diethyl phthalate, di-n-butyl phthalate, di-n-octyl phthalate, acenaphthylene, hexachlorobutadiene, and 2-methylphenol) did not exceed the SQS or the CSL.

Total PCB concentrations in surface sediment exceeded the SQS more frequently than any other chemical (37% of the sampled locations exceeded the SQS, and, of those, 14% also exceeded the CSL). The chemical with the next highest number of locations with detected concentrations in surface sediment exceeding the SQS was BEHP (12.9% of the sampled locations exceeded the SQS, and, of those, 7.1% also exceeded the CSL), followed by butyl benzyl phthalate (BBP) (9.6% of the sampled locations exceeded the SQS, and, of those, approximately 1% also exceeded the CSL) (Table 4-16). The remaining chemicals had SQS exceedance frequencies of 5.1% or less. Of the chemicals that infrequently exceeded the SQS in localized areas, those with the highest concentrations relative to the SQS were metals (e.g., lead), individual PAH compounds (e.g., acenapthene) and some SVOCs (e.g., 1,4-dichlorobenzene).

Table 4-16. Comparison of LDW surface sediment chemical data to SMS chemical criteria

Chemical	No. of Locations	Detection Frequency		Sediment Chemistry							
				Detected Concentrations > SQS and ≤ CSL[a]		Detected Concentrations > CSL[a]		Locations with RLs > SMS Criteria[c]		Maximum Ratio of Detected Conc. to SQS[a]	Maximum Ratio of Detected Conc. to CSL[a]
		No. of Detects	%	No.	%[b]	No.	%[b]	> SQS and ≤ CSL	> CSL		
Metals											
Arsenic	852	794	93	5	0.59	9	1.1	0	0	19	12
Cadmium	838	584	70	2	0.24	11	1.3	0	0	24	18
Chromium	850	850	100	1	0.12	8	0.94	0	0	4.2	4.1
Copper	852	852	100	0	0	12	1.4	0	0	31	31
Lead	852	852	100	2	0.23	19[d]	2.2	0	0	51	43
Mercury	868	746	86	16	1.8	27	3.1	0	0	11	7.8
Silver	823	499	61	0	0	10	1.2	0	0	44	44
Zinc	849	849	100	26	3.1	16	1.9	0	0	24	10
PAHs											
2-Methylnaphthalene	818	139	17	0	0	3	0.37	6	1	4.2	2.5
Acenaphthene	828	304	37	15	1.8	3	0.36	9	2	16	4.6
Acenaphthylene	818	128	16	0	0	0	0	0	1	0.38	0.38
Anthracene	828	576	70	2	0.24	0	0	0	0	1.7	0.32
Benzo(a)anthracene	828	748	90	9	1.1	4	0.48	0	0	4.0	1.6
Benzo(a)pyrene	822	747	91	5	0.61	4	0.49	0	0	4.2	2.0
Benzo(g,h,i)perylene	823	682	83	10	1.2	8	0.97	4	1	5.8	2.3
Total benzofluoranthenes	822	757	92	5	0.61	5	0.61	0	0	3.9	2.0
Chrysene	828	773	93	25	3.0	2	0.24	0	0	3.7	1.3
Dibenzo(a,h) anthracene	828	436	53	16	1.9	4	0.48	10	6	5.9	2.2
Dibenzofuran	827	248	30	6	0.73	3	0.36	9	2	15	3.8

Table 4-16, cont. Comparison of LDW surface sediment chemical data to SMS chemical criteria

Chemical	No. of Locations	Detection Frequency		Sediment Chemistry							
				Detected Concentrations > SQS and ≤ CSL[a]		Detected Concentrations > CSL[a]		Locations with RLs > SMS Criteria[c]		Maximum Ratio of Detected Conc. to SQS[a]	Maximum Ratio of Detected Conc. to CSL[a]
		No. of Detects	%	No.	%[b]	No.	%[b]	> SQS and ≤ CSL	> CSL		
Fluoranthene	828	797	96	32	3.9	10	1.2	0	0	8.1	2.7
Fluorene	828	382	46	10	1.2	3	0.36	8	1	13	3.7
Indeno(1,2,3-cd)pyrene	823	726	88	16	1.9	9	1.1	1	0	5.9	2.3
Naphthalene	818	152	19	0	0	2	0.24	0	0	2.6	1.7
Phenanthrene	828	759	92	24	2.9	3	0.36	0	0	15	3.1
Pyrene	828	788	95	2	0.24	4	0.48	0	0	1.8	1.5
Total HPAH[e]	828	803	97	24	2.9	4	0.48	0	0	4.7	1.5
Total LPAH[f]	828	763	92	3	0.36	3	0.36	0	0	6.2	2.9
Phthalates											
BEHP	832	674	81	48	5.8	59	7.1	3	2	11	7.4
BBP	822	425	52	71	8.6	8	0.97	75	1	110	8.3
Diethyl phthalate	832	41	4.9	0	0	0	0	4	0	0.33	0.18
Dimethyl phthalate	822	156	19	0	0	1	0.12	8	5	2.5	1.1
Di-n-butyl phthalate	822	189	23	0	0	0	0	0	0	0.64	0.082
Di-n-octyl phthalate	832	49	5.9	0	0	0	0	5	0	0.60	0.0078
Other SVOCs											
1,2,4-Trichlorobenzene	816	5	0.61	0	0	1	0.12	231	128	2.3	1.4
1,2-Dichlorobenzene	816	17	2.1	0	0	3	0.37	0	110	15	10
1,4-Dichlorobenzene	816	39	4.8	0	0	3	0.37	77	18	21	13
2,4-Dimethylphenol	813	5	0.62	0	0	1	0.12	0	222	10	10
2-Methylphenol	821	7	0.85	0	0	0	0	0	114	0.92	0.92
4-Methylphenol	831	82	9.9	0	0	4	0.48	0	9	6.9	6.9
Benzoic acid	822	70	8.5	0	0	8	0.97	0	107	6.9	6.9

Table 4-16, cont. Comparison of LDW surface sediment chemical data to SMS chemical criteria

		Detection Frequency		Sediment Chemistry							
				Detected Concentrations > SQS and ≤ CSL[a]		Detected Concentrations > CSL[a]		Locations with RLs > SMS Criteria[c]		Maximum Ratio of Detected Conc. to SQS[a]	Maximum Ratio of Detected Conc. to CSL[a]
Chemical	No. of Locations	No. of Detects	%	No.	%[b]	No.	%[b]	> SQS and ≤ CSL	> CSL		
Benzyl alcohol	812	15	1.8	2	0.25	3	0.37	7	106	12	9.2
Hexachlorobenzene	819	46	5.6	4	0.49	2	0.24	283	105	10	1.7
Hexachlorobutadiene	818	0	0	0	0	0	0	58	88	na	na
n-Nitrosodiphenylamine	818	23	2.8	0	0	2	0.24	34	33	3.9	2.8
Pentachlorophenol	785	12	1.5	1	0.13	0	0	88	29	1.1	0.59
Phenol	831	257	31	17	2.0	6	0.72	5	0	6.7	2.3
PCBs								0			
Total PCBs	1,327	1,243	94	314	24	174	13	0	0	830	150

[a] For individual locations with TOC ≥ 4% or ≤0.5%, locations were counted as greater than the SQS or CSL if the dry-weight concentration of chemicals with OC-normalized SMS criteria was greater than the LAET or 2LAET, respectively.

[b] Percentage of total number of locations with detected concentrations that were greater than the SQS and less than or equal to the CSL or greater than the CSL.

[c] Number of locations with RLs greater than the SQS or CSL.

[d] An additional sample collected from the Boeing Plant 2/Jorgensen Forge EAA had a lead concentration of 1,300 mg/kg dw, exceeding the CSL. According to data management rules (Section 4.1.2.1), data from this sample were excluded from the baseline RI dataset and replaced with data from a more recent sample (SD-343) collected within 10 ft of the original location, even though the more recent sample was not analyzed for lead.

[e] Total HPAHs were calculated as the sum of benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, total benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, and pyrene.

[f] Total LPAHs were calculated as the sum of acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
CSL – cleanup screening level
dw – dry weight
EAA – early action area
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LAET – lowest apparent effects threshold
2LAET – second lowest apparent effects threshold
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
na – not applicable
OC – organic carbon
PCB – polychlorinated biphenyl
RI – remedial investigation
RL – reporting limit
SMS – Washington State Sediment Management Standards
SQS – sediment quality standards
SVOC – semivolatile organic compound

Maps 4-14a through 4-14f present detailed results for each location with SQS or CSL exceedances for any detected chemical in surface sediment. Details on the locations of SQS and CSL exceedances for specific chemicals are provided in Sections 4.2.3 through 4.2.9.

For certain chemicals, analytical laboratory RLs exceeded the SQS (and in fewer cases, the CSL) (Table 4-16). These elevated RLs, which generally reflected dilutions or analytical challenges, create some uncertainty in interpreting the data relative to potential effects on benthic invertebrates. Appendix A, Section A.6.1.1.1, presents a detailed discussion of RL exceedances at all locations, including those with detected chemical exceedances.

The RL for at least one chemical was greater than the SQS at 294 locations with no detected exceedances of the SQS, including only two locations from sampling events conducted for the RI (Map 4-15). The increased sensitivity of the analytical methods used during the RI greatly reduced the frequency of RL values above the SQS. There was no increase in the detection frequency of these compounds, which suggests that these chemicals are likely not present in the LDW at concentrations above the RL.

Of the 294 locations with RLs greater than the SQS and no detected exceedances, 94% of the locations had RLs for 1,2,4-trichlorobenzene, hexachlorobenzene, or 2,4-dimethylphenol that were greater than the SQS (Map 4-15). As discussed in Section A.6.1.1 of the ERA, 1,2,4-trichlorobenzene, hexachlorobenzene, and 2,4-dimethylphenol are difficult to analyze at low concentrations using the standard SVOC analytical method and were rarely detected in LDW sediments (Table 4-16). Overall, approximately 44, 50, and 27% of all the surface sediment samples in which 1,2,4-trichlorobenzene, hexachlorobenzene, and 2,4-dimethylphenol, respectively, were not detected had RLs greater than the SQS. Detected concentrations of these compounds were greater than the SQS in one sample for both 1,2,4-trichlorobenzene and 2,4-dimethylphenol (out of five detected concentrations for each) and in six samples (out of 46 detected concentrations) for hexachlorobenzene.

There were 18 locations with no detected exceedances of the SQS but with RLs greater than the SQS for one or more chemicals other than 1,2,4-trichlorobenzene, hexachlorobenzene, and 2,4-dimethylphenol (Map 4-15). These other chemicals included: benzoic acid (1 location), benzyl alcohol (1 location), hexachlorobutadiene (9 locations; never detected in LDW), n-nitrosodiphenylamine (14 locations; 2 of 23 detected concentrations were greater than the SQS), pentachlorophenol (1 location), phenol (1 location), one individual PAH (1 location), and two phthalate compounds (1 location each). Benzoic acid, benzyl alcohol, hexachlorobutadiene, n-nitrosodiphenylamine, pentachlorophenol, and phenol are also difficult to analyze at low concentrations. Thus, the available data suggest that locations with RLs greater than the SQS but no detected exceedances are likely the result of analytical difficulties; the likelihood of actual chemical concentrations greater than the SQS at these locations is low.

4.2.1.3 Toxicity test results relative to SMS

Surface sediment samples collected during Rounds 1 and 2 of the RI were tested for toxicity if chemical concentrations exceeded the SQS or were greater than the screening level (SL) of the Dredged Material Management Program (DMMP) with the following exceptions:

- Ten samples had chemical concentrations that exceeded the SQS or were greater than the SL but were not tested because they were assumed to be toxic or were located in an EAA.
- Two samples were tested at the request of EPA/Ecology because they were near potential upland source areas; chemical concentrations in these samples did not exceed the SQS and they were less than the SL.

Of the 46 samples in the RI baseline dataset that were tested for toxicity during the RI, 11 sediment samples exceeded the overall SQS biological effects criteria (i.e., the SQS is exceeded for any one of the three endpoints), 17 sediment samples exceeded the overall CSL biological effects criteria (i.e., the CSL is exceeded for any one of the three endpoints or the SQS is exceeded for any two of the three endpoints), and 18 sediment samples did not exceed either the SQS or CSL biological effects criteria (Table 4-17; Map 4-13). Two samples, as noted above, were tested at the request of EPA and Ecology although they had no chemical SQS or CSL exceedances; these samples did not exceed either the SQS or CSL biological effects criteria. For the amphipod mortality endpoint, 10 of the 46 samples failed the biological effects criteria of the SMS at the CSL level, and 4 of the 46 samples failed the biological effects criteria at the SQS level (Table 4-17). For the polychaete growth endpoint, 8 of the 46 samples failed the biological effects criteria at the SQS level; no samples failed the biological effects criteria at the CSL level. For the bivalve survival/development endpoint, 8 of the 46 samples failed the biological effects criteria at the CSL level, and 12 of the 46 samples failed the biological effects criteria at the SQS level.

Maps 4-13 and 4-14a through-4-14f present both the toxicity and chemistry SMS exceedances for 46 locations where split sediment samples were analyzed for both toxicity and chemistry. As described previously in this section, toxicity test results, relative to SQS and CSL biological criteria, overrule SQS or CSL designations based on chemistry because the toxicity test results provide a more direct assessment of sediment toxicity at the location tested. The SMS designation based on toxicity test results agreed with the SMS designation based on chemistry for 17 of the 46 samples (Map 4-13). The toxicity designation indicated a higher quality condition (i.e., lower toxicity than estimated based on chemistry alone) at 20 of the 29 locations where the toxicity and chemistry designations did not agree and a lower quality condition (i.e., higher toxicity than estimated based on chemistry alone) at the remaining 9 locations (see Appendix A, Section A.3.2.2, for a comprehensive discussion of the toxicity test results).

Table 4-17. Results of chemistry and site-specific toxicity testing of surface sediment samples from the LDW

		Toxicity Test Results							
	Chemical Results	Amphipod Toxicity Test		Polychaete Toxicity Test			Bivalve Larval Toxicity Test		
Sample ID	SMS Exceedance	Percent Mean Mortality ± SD	SMS Exceedance[a, b]	Mean Mortality ± SD	Mean Individual Growth Rate ± SD (mg/day)	SMS Exceedance[a, c, d]	Percent Mean Normal Survivorship ± SD	SMS Exceedance[a, e]	Overall SMS Exceedance
LDW-SS2-010	SQS	39.0 ± 16.0	CSL	0.0 ± 0.0	0.76 ± 0.20	no exceedances	31.8 ± 15.1	CSL	CSL
LDW-SS6-010	CSL	47.0 ± 21.7	CSL	4.0 ± 8.9	0.81 ± 0.04	no exceedances	23.6 ± 16.3	CSL	CSL
LDW-SS15-010	CSL	28.0 ± 8.4	SQS	0.0 ± 0.0	0.73 ± 0.10	no exceedances	75.4 ± 8.7	no exceedances	SQS
LDW-SS16-010	SQS	16.0 ± 10.2	no exceedances	4.0 ± 8.9	0.96 ± 0.13	no exceedances	64.3 ± 3.2	SQS	SQS
LDW-SS17-010	SQS	35.0 ± 20.9	SQS	0.0 ± 0.0	0.92 ± 0.22	no exceedances	62.9 ± 7.6	no exceedances	SQS
LDW-SS24-010	CSL	7.0 ± 4.5	no exceedances	0.0 ± 0.0	0.77 ± 0.17	SQS	18.3 ± 3.3	CSL	CSL
LDW-SS26-010	SQS	23.0 ± 11.0	no exceedances	0.0 ± 0.0	0.86 ± 0.17	no exceedances	76.7 ± 7.4	no exceedances	no exceedances
LDW-SS29-010	no exceedances	12.0 ± 7.6	no exceedances	0.0 ± 0.0	0.90 ± 0.08	no exceedances	64.7 ± 6.8	no exceedances	no exceedances
LDW-SS31-010	CSL	43.0 ± 5.7	CSL	4.0 ± 8.9	0.81 ± 0.10	no exceedances	62.9 ± 7.6	SQS	CSL
LDW-SS32-010	SQS	34.0 ± 11.9	SQS	0.0 ± 0.0	0.69 ± 0.10	no exceedances	78.9 ± 15.7[f]	no exceedances	SQS
LDW-SS37-010	CSL	45.0 ± 14.6	CSL	0.0 ± 0.0	0.78 ± 0.05	no exceedances	65.8 ± 19.2[g]	SQS	CSL
LDW-SS39-010	CSL	29.0 ± 11.4	SQS	0.0 ± 0.0	0.76 ± 0.12	SQS	83.4 ± 10.1	no exceedances	CSL
LDW-SS40-010	SQS	36.0 ± 12.9	CSL	0.0 ± 0.0	0.78 ± 0.14	no exceedances	79.7 ± 4.9	no exceedances	CSL
LDW-SS49-010	CSL	49.0 ± 19.5	CSL	0.0 ± 0.0	0.79 ± 0.22	no exceedances	55.1 ± 17.4	SQS	CSL
LDW-SS50-010	SQS	39.0 ± 10.8	CSL	0.0 ± 0.0	0.73 ± 0.11	no exceedances	70.2 ± 10.2	no exceedances	CSL
LDW-SS56-010	CSL	6.0 ± 4.2	no exceedances	0.0 ± 0.0	0.85 ± 0.14	no exceedances	67.6 ± 7.2	SQS	SQS
LDW-SS57-010	SQS	13.0 ± 12.5	no exceedances	0.0 ± 0.0	0.78 ± 0.13	no exceedances	55.3 ± 15.1	CSL	CSL
LDW-SS58-010	SQS	5.0 ± 5.0	no exceedances	0.0 ± 0.0	0.69 ± 0.07	SQS	61.0 ± 4.0	SQS	CSL
LDW-SS60-010	SQS	7.0 ± 5.7	no exceedances	4.0 ± 8.9	0.77 ± 0.17	no exceedances	84.7 ± 6.0	no exceedances	no exceedances
LDW-SS63-010	no exceedances	5.0 ± 6.1	no exceedances	0.0 ± 0.0	0.68 ± 0.19	no exceedances	80.0 ± 1.6	no exceedances	no exceedances

Table 4-17, cont. Results of chemistry and site-specific toxicity testing of surface sediment samples from the LDW

		Toxicity Test Results							
	Chemical Results	Amphipod Toxicity Test		Polychaete Toxicity Test			Bivalve Larval Toxicity Test		
Sample ID	SMS Exceedance	Percent Mean Mortality ± SD	SMS Exceedance[a, b]	Mean Mortality ± SD	Mean Individual Growth Rate ± SD (mg/day)	SMS Exceedance[a, c, d]	Percent Mean Normal Survivorship ± SD	SMS Exceedance[a, e]	Overall SMS Exceedance
LDW-SS68-010	CSL	12.0 ± 9.1	no exceedances	0.0 ± 0.0	0.85 ± 0.10	no exceedances	71.6 ± 12.5	no exceedances	no exceedances
LDW-SS69b-010	SQS	37.0 ± 15.7	CSL	0.0 ± 0.0	0.85 ± 0.29	no exceedances	59.0 ± 13.6	SQS	CSL
LDW-SS70-010	SQS	15.0 ± 7.1	no exceedances	4.0 ± 8.9	0.78 ± 0.12	no exceedances	60.7 ± 12.0	SQS	SQS
LDW-SS71-010	SQS	5.0 ± 6.1	no exceedances	0.0 ± 0.0	0.92 ± 0.16	no exceedances	61.8 ± 8.7	no exceedances	no exceedances
LDW-SS73-010	CSL	12.0 ± 9.1	no exceedances	0.0 ± 0.0	0.86 ± 0.11	no exceedances	56.8 ± 13.3	SQS	SQS
LDW-SS75-010	SQS	8.0 ± 7.6	no exceedances	4.0 ± 8.9	0.69 ± 0.16	no exceedances	76.5 ± 7.5	no exceedances	no exceedances
LDW-SS77-010	SQS	16.0 ± 9.6	no exceedances	4.0 ± 8.9	0.95 ± 0.12	no exceedances	10.1 ± 4.0	CSL	CSL
LDW-SS85-010	SQS	1.0 ± 2.2	no exceedances	0.0 ± 0.0	0.87 ± 0.18	no exceedances	86.8 ± 5.3	no exceedances	no exceedances
LDW-SS88-010	CSL	48.0 ± 25.9	CSL	0.0 ± 0.0	0.68 ± 0.14	no exceedances	11.9 ± 5.3	CSL	CSL
LDW-SS89-010	CSL	5.0 ± 3.5	no exceedances	0.0 ± 0.0	0.84 ± 0.26	no exceedances	86.9 ± 5.1[g]	no exceedances	no exceedances
LDW-SS92-010	CSL	1.0 ± 2.2	no exceedances	0.0 ± 0.0	0.79 ± 0.17	no exceedances	78.3 ± 14.3	no exceedances	no exceedances
LDW-SS106-010	SQS	6.0 ± 4.2	no exceedances	0.0 ± 0.0	0.91 ± 0.11	no exceedances	61.4 ± 8.8	no exceedances	no exceedances
LDW-SS112-010	CSL	4.0 ± 4.2	no exceedances	0.0 ± 0.0	0.82 ± 0.06	no exceedances	65.4 ± 9.6	no exceedances	no exceedances
LDW-SS114-010	CSL	85.0 ± 7.1	CSL	4.0 ± 8.9	0.77 ± 0.19	no exceedances	56.6 ± 8.5	SQS	CSL
LDW-SS115-010	SQS	9.0 ± 4.2	no exceedances	0.0 ± 0.0	0.76 ± 0.21	no exceedances	77.6 ± 11.0	no exceedances	no exceedances
LDW-SS119-010	SQS	3.0 ± 2.7	no exceedances	0.0 ± 0.0	0.79 ± 0.06	no exceedances	68.8 ± 7.6	no exceedances	no exceedances
LDW-SS120-010	SQS	3.0 ± 2.7	no exceedances	0.0 ± 0.0	0.71 ± 0.09	no exceedances	56.3 ± 11.2	SQS	SQS
LDW-SS121-010	CSL	4.0 ± 4.2	no exceedances	0.0 ± 0.0	0.90 ± 0.12	no exceedances	76.4 ± 8.0	no exceedances	no exceedances
LDW-SS122-010	SQS	7.0 ± 4.5	no exceedances	0.0 ± 0.0	0.83 ± 0.14	no exceedances	68.5 ± 14.9	no exceedances	no exceedances
LDW-SS143-010	CSL	6.0 ± 5.5	no exceedances	0.0 ± 0.0	0.75 ± 0.07	no exceedances	72.8 ± 3.2	no exceedances	no exceedances
LDW-SS144-010	SQS	1.0 ± 2.2	no exceedances	0.0 ± 0.0	0.72 ± 0.11	SQS	66.4 ± 12.1	no exceedances	SQS[h]

Table 4-17, cont. Results of chemistry and site-specific toxicity testing of surface sediment samples from the LDW

	CHEMICAL RESULTS	TOXICITY TEST RESULTS							
		AMPHIPOD TOXICITY TEST		POLYCHAETE TOXICITY TEST			BIVALVE LARVAL TOXICITY TEST		
SAMPLE ID	SMS EXCEEDANCE	PERCENT MEAN MORTALITY ± SD	SMS EXCEEDANCE[a, b]	MEAN MORTALITY ± SD	MEAN INDIVIDUAL GROWTH RATE ± SD (mg/day)	SMS EXCEEDANCE[a, c, d]	PERCENT MEAN NORMAL SURVIVORSHIP ± SD	SMS EXCEEDANCE[a, e]	OVERALL SMS EXCEEDANCE
LDW-SS148-010	SQS	6.0 ± 6.5	no exceedances	0.0 ± 0.0	0.78 ± 0.08	SQS	29.9 ± 6.6	CSL	CSL
LDW-SS157-010	CSL	8.0 ± 7.6	no exceedances	4.0 ± 8.9	0.78 ± 0.14	SQS	71.6 ± 8.5	no exceedances	SQS[h]
LDW-SS158-010	SQS	12.0 ± 4.5	no exceedances	0.0 ± 0.0	0.81 ± 0.10	no exceedances	67.5 ± 6.5	no exceedances	no exceedances
LDW-SSB2b-010	SQS	25.0 ± 12.2	no exceedances	0.0 ± 0.0	1.02 ± 0.10	no exceedances	42.1 ± 20.0	CSL	CSL
LDW-SSB6a-010	SQS	2.0 ± 4.5	no exceedances	0.0 ± 0.0	0.82 ± 0.14	SQS	60.1 ± 13.3	no exceedances	SQS[h]

[a] Statistical analyses in SEDQUAL 5.0 included Wilk-Shapiro test for normality and Levene's test for equality of variances followed by the appropriate statistical test for significance (i.e., Student's t-test, approximate t-test, or Mann-Whitney test).

[b] SQS – mean mortality > 25% on an absolute basis and statistically different from the reference sediment ($p \leq 0.05$); CSL – mean mortality greater than the value in the reference sediment plus 30% and statistically different from the reference sediment ($p \leq 0.05$). Reference sediment results are presented in the Round 1 and Round 2 surface sediment data reports (Windward 2005c, d).

[c] SQS – mean individual growth rate <70% of that of the reference sediment and statistically different ($p \leq 0.05$).

[d] No exceedance was reported for the polychaete growth endpoint for some of the sediment samples because of high variability in the reference and/or test samples.

[e] SQS – mean normal survivorship < 85% of that of the reference sediment and statistically different ($p \leq 0.10$); CSL – mean normal survivorship < 70% of that of the reference sediment and statistically different ($p \leq 0.10$).

[f] One of the five replicates from each of these tests was double-inoculated, so those replicates were not used in calculating mean normal survivorship and mean effective mortality for those test sediments.

[g] One of the five replicates from each of these tests was not inoculated at test initiation, so those replicates were not used in calculating mean normal survivorship and mean effective mortality for those test sediments.

[h] Exceeded the SQS criterion based on reduction in polychaete growth alone.

CSL – cleanup screening level
ID – identification
LDW – Lower Duwamish Waterway
SMS – Washington State Sediment Management Standards
SD – standard deviation
SQS – sediment quality standards

Thiessen polygons were used to estimate the areal extent of potential effects based on combined toxicity test results and surface sediment chemistry data (see Map 4-16). Using this approach, approximately 75% of the LDW area (345 ac) had chemical concentrations less than or equal to chemical SQS criteria, and sediments were non-toxic according to biological SQS criteria. In approximately 7% of the LDW area (34 ac), chemical concentrations or biological effects exceeded CSL criteria. The remaining 18% of the LDW area (82 ac) had chemical concentrations or biological effects between the SQS and CSL criteria. These area estimates are uncertain because they were based on interpolation using Thiessen polygons.

4.2.1.4 Subsurface sediment chemistry relative to SMS

Table 4-18 presents a summary of subsurface sediment samples with chemical concentrations greater than the SQS and CSL. Detected concentrations of 36 SMS chemicals were greater than the CSL at one or more locations. Eight additional chemicals (i.e., anthracene, benzo(g,h,i)perylene, indeno(1,2,3-cd)pyrene, butyl benzyl phthalate, di-n-octyl phthalate, hexachlorobenzene, hexachlorobutadiene, and n-nitrosodiphenylamine) were detected at concentrations that exceeded the SQS but did not exceed the CSL. Detected concentrations of the remaining three SMS chemicals (i.e., acenapthylene, di-n-butyl phthalate, and 4-methylphenol) did not exceed the SQS or the CSL.

Maps 4-17a through 4-17j show the distribution of chemicals with concentrations greater than the SMS criteria in subsurface samples outside of the Duwamish/Diagonal, Boeing Plant 2, and T-117 EAAs;[54] these maps show exceedances for specific chemical groups (i.e., PCBs, BEHP, SVOCs [excluding BEHP], and metals). Cores collected from the portion of the LDW between RM 0.0 and RM 1.5 frequently had subsurface samples with detected concentrations that were greater than the SQS and the CSLs. Most of these samples had PCB concentrations greater than the SQS (Map 4-17a), but there were also a number of other chemicals with concentrations greater than the SQS. These chemicals were primarily mercury, BEHP, and other metals (Map 4-17b).

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[54] These areas were not included on Maps 4-16a through 4-16j because of the substantial number of cores collected and because these data will be evaluated in greater detail as part of individual site investigations.

Table 4-18. Comparison of subsurface sediment data to SMS criteria

Analyte	n[a]	Detection Frequency		Subsurface Sediment Chemistry							
				Detected Concentrations > SQS and ≤ CSL[b]		Detected Concentrations > CSL[b]		Samples with RLs > SMS Criteria[c]		Maximum Ratio ofDetected Conc. to SQS	Maximum Ratio of Detected Conc. to CSL
		No. of Detects	%	No.	%[d]	No.	%[d]	> SQS and ≤ CSL	CSL		
Metals and Trace Elements											
Arsenic	325	267	82	6	1.8	19	5.8	0	0	35	22
Cadmium	388	257	66	1	0.26	8	2.1	0	0	4.0	3.0
Chromium	397	397	100	0	0	2	0.50	0	0	1.5	1.4
Copper	397	397	100	0	0	10	2.5	0	0	7.5	7.5
Lead	400	367	92	3	0.75	10	2.5	0	0	7.8	6.6
Mercury	440	345	78	22	5.0	33	7.5	0	0	24	17
Silver	372	166	45	0	0	2	0.54	0	0	1.2	1.2
Zinc	396	396	100	15	3.8	15	3.8	0	0	12	4.9
PAHs											
2-Methylnaphthalene	297	46	15	0	0	1	0.34	0	0	6.7	3.2
Acenaphthene	320	104	33	11	3.4	4	1.3	3	0	10	6.3
Acenaphthylene	297	57	19	0	0	0	0	0	0	0.22	0.22
Anthracene	320	216	68	2	0.63	0	0	0	0	3.1	0.57
Benzo(a)anthracene	320	259	81	6	1.9	2	0.63	0	0	5.0	2.3
Benzo(a)pyrene	320	253	79	4	1.3	2	0.63	0	0	2.3	1.1
Benzo(g,h,i)perylene	320	222	69	7	2.2	0	0	2	0	1.8	0.82
Total benzofluoranthenes	320	264	83	4	1.3	2	0.63	0	0	2.4	2.1
Chrysene	320	263	82	8	2.5	2	0.63	0	0	5.5	1.5
Dibenzo(a,h)anthracene	320	135	42	21	6.6	1	0.31	1	5	3.2	1.2
Dibenzofuran	320	66	21	7	2.2	2	0.63	4	0	6.7	2.4
Fluoranthene	320	278	87	16	5.0	3	0.94	0	0	11	3.2
Fluorene	320	112	35	4	1.3	3	0.94	2	0	8.0	4.3
Indeno(1,2,3-cd)pyrene	320	226	71	9	2.8	0	0	0	0	2.1	0.97

Table 4-18, cont. Comparison of subsurface sediment data to SMS criteria

Analyte	n[a]	Subsurface Sediment Chemistry									
		Detection Frequency		Detected Concentrations > SQS and ≤ CSL[b]		Detected Concentrations > CSL[b]		Samples with RLs > SMS Criteria[c]		Maximum Ratio of Detected Conc. to SQS	Maximum Ratio of Detected Conc. to CSL
		No. of Detects	%	No.	%[d]	No.	%[d]	> SQS and ≤ CSL	CSL		
Naphthalene	309	75	24	0	0	1	0.32	0	0	1.6	1.4
Phenanthrene	320	260	81	6	1.9	2	0.63	0	0	9.2	2.4
Pyrene	320	282	88	1	0.31	2	0.63	0	0	2.6	2.0
Total HPAH	320	284	89	9	2.8	2	0.63	0	0	5.1	2.0
Total LPAH	320	260	81	3	0.94	2	0.63	0	0	5.2	2.4
Phthalates											
BEHP	322	231	72	27	8.4	19	5.9	6	5	4.3	2.6
Butyl benzyl phthalate	299	163	55	17	5.7	0	0	7	0	5.8	0.41
Diethyl phthalate	322	1	0.31	0	0	1	0.31	1	0	6.7	3.7
Dimethyl phthalate	299	20	6.7	0	0	1	0.33	2	2	6.4	6.4
Di-n-butyl phthalate	299	64	21	0	0	0	0	0	0	0.086	0.024
Di-n-octyl phthalate	322	14	4.3	1	0.31	0	0	5	0	2.4	0.031
Other SVOCs											
1,2,4-Trichlorobenzene	309	29	9.4	3	0.97	2	0.65	49	8	3.5	2.2
1,2-Dichlorobenzene	308	35	11	0	0	3	0.97	1	7	4.3	4.3
1,4-Dichlorobenzene	308	60	19	0	0	1	0.32	2	2	6.8	6.3
2,4-Dimethylphenol	297	22	7.4	0	0	1	0.34	0	16	1.6	1.6
2-Methylphenol	297	38	13	0	0	1	0.34	0	4	2.5	2.5
4-Methylphenol	320	15	4.7	0	0	0	0	0	0	0.16	0.16
Benzoic acid	320	122	38	0	0	13	4.1	0	16	4.6	4.6
Benzyl alcohol	297	33	11	2	0.67	6	2.0	14	11	3.7	2.9
Hexachlorobenzene	296	3	1.0	1	0.34	0	0	127	7	1.8	0.30
Hexachlorobutadiene	309	2	0.65	1	0.32	0	0	8	4	1.2	0.11
n-Nitrosodiphenylamine	297	1	0.34	1	0.34	0	0	5	19	1.2	0.83
Pentachlorophenol	320	55	17	1	0.31	4	1.3	2	23	2.6	1.3

Table 4-18, cont. Comparison of subsurface sediment data to SMS criteria

				SUBSURFACE SEDIMENT CHEMISTRY							
		DETECTION FREQUENCY		DETECTED CONCENTRATIONS > SQS AND ≤ CSL[b]		DETECTED CONCENTRATIONS > CSL[b]		SAMPLES WITH RLs > SMS CRITERIA[c]		MAXIMUM RATIO OFDETECTED CONC. TO SQS	MAXIMUM RATIO OF DETECTED CONC. TO CSL
ANALYTE	n[a]	NO. OF DETECTS	%	NO.	%[d]	NO.	%[d]	> SQS AND ≤ CSL	CSL		
Phenol	320	57	18	1	0.31	1	0.31	1	0	7.4	2.6
PCBs											
PCBs (total calc'd)	871	654	75	229	26	184	21	3	0	2,400	450

[a] The total number of samples includes all analyzed core intervals and does not represent the total number of cores.

[b] For individual samples with TOC ≥ 4.0% or ≤ 0.5%, samples were counted as greater than the SQS or CSL if the dry-weight concentration was greater than the LAET or 2LAET, respectively.

[c] Number of samples with RLs greater than the SQS or CSL, when undetected.

[d] Percentage of total number of locations with detected concentrations greater than the SQS or CSL. Percent exceeding SQS excludes samples with CSL exceedances.

BEHP – bis(2-ethylhexyl) phthalate
CSL – cleanup screening level
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
n – number of samples
PCB – polychlorinated biphenyl
SMS – Washington State Sediment Management Standards
SQS – sediment quality standards
SVOC – semivolatile organic compound

Between RM 1.5 and RM 4.0, there were comparatively fewer samples with chemical concentrations greater than the CSL, but samples frequently had concentrations greater than the SQS (Maps 4-17c through 4-17h). Most of the concentrations greater than the SQS were for total PCBs. South of RM 4.0, most of the concentrations greater than the SQS in subsurface samples were located in the vicinity of the Rhône-Poulenc site, including Slip 6 (RM 4.0 to RM 4.2), or in the vicinity of the Norfolk EAA (Maps 4-17i and 4-17j). Chemicals most frequently detected in subsurface samples at concentrations greater than the SQS in the vicinity of the Slip 6 were benzoic acid and dibenzo(a,h)anthracene. Chemicals most frequently detected in subsurface samples at concentrations greater than the SQS in the vicinity of the Norfolk EAA were PCBs, 1,4-dichlorobenzene, BEHP, and mercury. The cores in which these exceedances were detected (NFK207, NFK008, and NFK009) no longer represent current conditions because those core locations were subsequently dredged.

4.2.2 Water chemistry relative to water quality criteria

The LDW site includes the water column as well as the sediments. Chemical concentrations in biota are affected by chemicals in surface water as well as sediments, as discussed in Appendix D. This section describes the numerical criteria for toxic pollutants in surface water and summarizes LDW surface water, seep water, and porewater data, as requested by EPA and Ecology.

Federal water quality criteria (WQC) developed to protect ecological receptors and human consumers of fish and shellfish are relevant and appropriate requirements or minimum levels or standards for remedial action pursuant to CERCLA Section 121 (d)(2)(A)(ii) and RCW 70.105D.030(2)(e). Under CERCLA and MTCA, state water quality standards (WQS) approved by EPA are generally applicable requirements under the Clean Water Act (CWA). National recommended federal WQC established pursuant to Section 304(a)(1) of the CWA are compiled and presented on the EPA website at http://www.epa.gov/waterscience/criteria/wqctable/. Although these criteria are advisory only for CWA purposes, the last sentence of CERCLA Section 121 (d)(2)(A)(ii) makes them minimum cleanup levels or standards for the site. Consequently, the more stringent of the federal WQC and the state WQS approved by EPA for CWA purposes are the cleanup levels or standards for the site.

Washington State WQS for the protection of aquatic life are found at WAC 173-201A-240. The numerical criteria for aquatic life meet the federal requirements of Section 303(c)(2)(B) of the CWA, are at least as stringent as the federal WQC, and have been approved by EPA as applicable CWA criteria in accordance with 40 CFR 131.21.[55] The criteria for human health are listed in EPA's regulations at 40 CFR 131.36(d)(14) and

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[55] Except for the Washington State copper criteria, which were amended by Washington in 2007 (72 Fed. Reg. 37,109, 37,115 [July 9, 2007]) and approved by EPA (40 CFR 131.36). Federal WQC are also presented in the tables in this section if their values are lower than state WQC.

cross-referenced in Washington's regulations at WAC 173-201A-240(5). The federal and state aquatic life and human health WQS and WQC are hereafter referred to as WQC in this document unless otherwise noted.

The aquatic life WQC are presented as both acute and chronic criteria. Aquatic life WQC are based on dissolved concentrations for metals (except mercury) and total concentrations for mercury and organic compounds. Acute WQC are 1-hr average concentrations not to be exceeded more than once every 3 years, with the exception of silver and pesticide concentrations, which are instantaneous concentrations not to be exceeded at any time, or the PCB concentration, which is a 24-hr average not to be exceeded at any time. Chronic WQC are 4-day average concentrations not to be exceeded more than once every 3 years, with the exception of pesticide and PCB concentrations, which are 24-hr average concentrations not to be exceeded at any time.

Both marine and freshwater values have been developed for aquatic life WQC. Under state regulations, freshwater values apply to waters with a salinity of 1 ppt or less 95% of the time (WAC 173-201A-260). These regulations also state that in brackish water of estuaries, where different criteria for the same use occurs for fresh and marine waters, the decision to use the freshwater or the marine water criteria must be selected and applied on the basis of vertically averaged daily maximum salinity. Federal regulations (40 CFR 131.36[c][3]) state that for waters in which the salinity is between 1 and 10 ppt, the applicable criteria are the more stringent of the freshwater or saltwater criteria. They do not contain provisions for vertically averaging salinity data.

As noted in Section 2.7, LDW is a well-stratified estuary. Fresh water flowing from the Green/Duwamish River system enters the headwaters of the LDW, and salt water from Puget Sound enters the lower reaches of the LDW from the mouth of the waterway. Typical of tidally influenced estuaries, the LDW has a relatively sharp interface between the freshwater outflow at the surface and saltwater inflow at depth. As the fresh water flows over the deeper saltwater wedge, only limited mixing occurs between these freshwater and saltwater lenses, resulting in a lens of fresh water overlying the salt wedge over a significant portion of the LDW a significant portion of the time. The salinity of the surface water varies with river flow and tidal conditions; during times of high river flow, the salinity in the surface water is low; whereas, during low-flow conditions, the surface water salinity is higher. At the river's mouth at the northern end of Harbor Island, a salinity of 25 ppt is typical for the entire water column; salinity decreases toward the upriver portion of the estuary. The upstream location or "toe" of the saltwater wedge, typically located between Slip 4 and the head of the navigation channel, is determined by both tidal elevation and freshwater inflow. As noted in Section 2.8, a more diverse biological community typical of marine environments generally exists in the downstream, more saline portion of the LDW (RM 0.0 to RM 2.0), transitioning upstream to a less-diverse community of species more tolerant of fluctuations in salinity (see Section 2.8.2.2).

Absent site-specific biological data to the contrary as outlined below, applicable (state) or relevant and appropriate (federal) criteria for LDW surface water data are the more stringent of state or federal WQC. In areas where the salinity in any portion or lens of the water column is greater than 1 ppt and less than 10 ppt 95% of the time, the more stringent of marine or freshwater criteria will be used. In areas where the salinity in all portions of the water column is greater than 10 ppt 95% of the time, only marine standards will be used. Freshwater criteria shall presumptively be used for any fresh water in the LDW pursuant to CERCLA section 121(d)(2)(A)(ii), unless it is demonstrated that they are not appropriate at particular locations based on site-specific biological information demonstrating that there are no freshwater biological receptors of significance found or expected to be present at such location(s).

For all locations generally, the determination of which standards apply must be made based on area-specific salinity measurements. The salinity in some areas is highly variable, depending on the tide and the flow rate from the Green/Duwamish River. For example, based on monthly salinity data collected at RM 3.3 as part of the King County Monthly Marine Monitoring Program (http://green.kingcounty.gov/marine/CTD.aspx), the salinity at this location is generally greater than 1 ppt and less than 10 ppt in the surface layer, which varies in depth from < 0.5 to 5 meters, with the deeper depths occurring under high river flow conditions. The more stringent of the freshwater or marine criteria would apply to this surface layer, provided the salinity is in the 1 to10 ppt range 95% of the time. Both freshwater and marine WQC are provided in Tables 4-19 though 4-22.

Human health WQC have been established for the consumption of organisms only (i.e., fish and shellfish) and also for the consumption of both organisms and water. The WQC based on consumption of both organisms and water are assigned to waters classified only for domestic water supply (40 CFR 131.36 [14]) and thus do not apply to the LDW. Human health WQC are based on dissolved concentrations for all chemicals.

Tables 4-19 through 4-22 present the criteria as well as summaries of surface water, seep water, and porewater data from the LDW. This information is provided only for informational purposes; comparisons of some of these criteria with the data may not be appropriate.

Table 4-19. Aquatic life WQC and human health WQC

CHEMICAL	STATE WQC (µg/L)[a]				FEDERAL AWQC (µg/L)[b]				
	FRESHWATER[c]		MARINE[c]		FRESHWATER[c]		MARINE[c]		HUMAN HEALTH[d]
	ACUTE[e]	CHRONIC[f]	ACUTE[e]	CHRONIC[f]	ACUTE[e]	CHRONIC[f]	ACUTE[e]	CHRONIC[f]	ORGANISMS ONLY
Metals and Trace Elements									
Antimony	na	na	na	na	na	na	na	na	640
Arsenic	360	190	**69**	**36**	**340**	**150**	69	36	0.14[g,h]
Cadmium	3.7	1.0	42	9.3	**2.0**	**0.25**	**40**	**8.8**	na
Chromium (hexavalent)	**15**	**10**	**1,100**	**50**	16	11	1,100	50	na
Chromium (trivalent)	**550**	180	na	na	570	**74**	na	na	na
Copper	**17**	**11**	**4.8**	**3.1**	nc[j]	nc[j]	4.8	3.1	na
Lead	**65**	**2.5**	**210**	**8.1**	65	2.5	210	8.1	na
Mercury	2.1	**0.012**	**1.8**	**0.025**	**1.4**	0.77	1.8	0.94	0.15[i]
Nickel	1,400	160	**74**	**8.2**	**470**	**52**	74	8.2	4,600
Selenium	**20**	**5**	**290**	**71**	na	5	290	71	4,200
Silver	3.4	na	**1.9**	na	**3.2**	na	1.9	na	na
Thallium	na	na	na	na	na	na	na	na	0.47
Zinc	**110**	**100**	**90**	**81**	120	120	90	81	26,000
PAHs									
2-Chloronaphthalene	na	na	na	na	na	na	na	na	1,600
Acenaphthene	na	na	na	na	na	na	na	na	990
Anthracene	na	na	na	na	na	na	na	na	40,000
Benzo(a)anthracene	na	na	na	na	na	na	na	na	0.018[g]
Benzo(a)pyrene	na	na	na	na	na	na	na	na	0.018[g]
Benzo(b)fluoranthene	na	na	na	na	na	na	na	na	0.018[g]
Benzo(k)fluoranthene	na	na	na	na	na	na	na	na	0.018[g]
Chrysene	na	na	na	na	na	na	na	na	0.018[g]
Dibenzo(a,h)anthracene	na	na	na	na	na	na	na	na	0.018[g]
Fluoranthene	na	na	na	na	na	na	na	na	140

Table 4-19, cont. Aquatic life WQC and human health WQC

	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
Chemical	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
Fluorene	na	na	na	na	na	na	na	na	5,300
Indeno(1,2,3-cd)pyrene	na	na	na	na	na	na	na	na	0.018[g]
Pyrene	na	na	na	na	na	na	na	na	4,000
Phthalates									
BEHP	na	na	na	na	na	na	na	na	2.2[g]
Butyl benzyl phthalate	na	na	na	na	na	na	na	na	1,900
Diethyl phthalate	na	na	na	na	na	na	na	na	44,000
Dimethyl phthalate	na	na	na	na	na	na	na	na	1,100,000
Di-n-butyl phthalate	na	na	na	na	na	na	na	na	4,500
SVOCs									
1,2,4-Trichlorobenzene	na	na	na	na	na	na	na	na	70
1,2-Dichlorobenzene	na	na	na	na	na	na	na	na	1,300
1,2-Diphenylhydrazine	na	na	na	na	na	na	na	na	0.2[g]
1,3-Dichlorobenzene	na	na	na	na	na	na	na	na	960
1,4-Dichlorobenzene	na	na	na	na	na	na	na	na	190
2,4,6-Trichlorophenol	na	na	na	na	na	na	na	na	2.4[g]
2,4-Dichlorophenol	na	na	na	na	na	na	na	na	290
2,4-Dimethylphenol	na	na	na	na	na	na	na	na	850
2,4-Dinitrophenol	na	na	na	na	na	na	na	na	5,300
2,4-Dinitrotoluene	na	na	na	na	na	na	na	na	3.4[g]
2-Chlorophenol	na	na	na	na	na	na	na	na	150
3,3'-Dichlorobenzidine	na	na	na	na	na	na	na	na	0.028[g]
4,6-Dinitro-o-cresol (2-methyl-4,6-dinitrophenol)	na	na	na	na	na	na	na	na	280
Benzidine	na	na	na	na	na	na	na	na	0.0002
Bis(2-chloroethyl)ether	na	na	na	na	na	na	na	na	0.53[g]
Bis(2-chloroisopropyl)ether	na	na	na	na	na	na	na	na	65,000
Hexachlorobenzene	na	na	na	na	na	na	na	na	0.00029[g]

Table 4-19, cont. Aquatic life WQC and human health WQC

	STATE WQC (µg/L)[a]				FEDERAL AWQC (µg/L)[b]				
	FRESHWATER[c]		MARINE[c]		FRESHWATER[c]		MARINE[c]		HUMAN HEALTH[d]
CHEMICAL	ACUTE[e]	CHRONIC[f]	ACUTE[e]	CHRONIC[f]	ACUTE[e]	CHRONIC[f]	ACUTE[e]	CHRONIC[f]	ORGANISMS ONLY
Hexachlorobutadiene	na	na	na	na	na	na	na	na	18[g]
Hexachlorocyclopentadiene	na	na	na	na	na	na	na	na	1,100
Hexachloroethane	na	na	na	na	na	na	na	na	3.3[g]
Isophorone	na	na	na	na	na	na	na	na	960[g] (**600**[i])
Nitrobenzene	na	na	na	na	na	na	na	na	690
n-Nitrosodimethylamine	na	na	na	na	na	na	na	na	3[g]
n-Nitroso-di-n-propylamine	na	na	na	na	na	na	na	na	0.51[g]
n-Nitrosodiphenylamine	na	na	na	na	na	na	na	na	6[g]
Pentachlorophenol	20[k]	**13**[k]	**13**	**7.9**	**19**[k]	15[k]	13	7.9	3[g]
Phenol	na	na	na	na	na	na	na	na	860,000
PCBs									
PCBs	**2**	**0.014**	**10**	**0.03**	na	0.014	na	0.03	0.000064[g]
Pesticides									
4,4′-DDD	na	na	na	na	na	na	na	na	0.00031[g]
4,4′-DDE	na	na	na	na	na	na	na	na	0.00022[g]
4,4′-DDT	**1.1**	**0.001**	**0.13**	**0.001**	1.1	0.001	0.13	0.001	0.00022[g]
Aldrin	na	na	na	na	**3.0**	na	**1.3**	na	0.000050[g]
Dieldrin	na	na	na	na	**0.24**	**0.056**	**0.71**	**0.0019**	0.000054[g]
Aldrin/dieldrin (sum)[l]	**2.5**	**0.0019**	**0.71**	**0.0019**	na	na	na	na	na
alpha-BHC	na	na	na	na	na	na	na	na	0.0049[g]
beta-BHC	na	na	na	na	na	na	na	na	0.017[g]
gamma-BHC (Lindane)	2.0	**0.08**	**0.16**	na	**0.95**	na	0.16	na	1.8
alpha-Endosulfan	**0.22**[m]	**0.056**[m]	**0.034**[m]	**0.0087**[m]	0.22	0.056	0.034	0.0087	89 (**2**[i])
beta-Endosulfan	**0.22**[m]	**0.056**[m]	**0.034**[m]	**0.0087**[m]	0.22	0.056	0.034	0.0087	89 (**2**[i])
Endosulfan sulfate	na	na	na	na	na	na	na	na	89 (**2**[i])
Endrin	0.18	**0.0023**	**0.037**	**0.0023**	**0.086**	0.036	0.037	0.0023	0.06
Endrin aldehyde	na	na	na	na	na	na	na	na	0.3

Table 4-19, cont. Aquatic life WQC and human health WQC

Chemical	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
Heptachlor	**0.52**	**0.0038**	**0.053**	**0.0036**	0.52	0.0038	0.053	0.0036	0.000079[g]
Heptachlor epoxide	na	na	na	na	**0.52**	**0.0038**	**0.053**	**0.0036**	0.000039[g]
Toxaphene	**0.73**	**0.0002**	**0.21**	**0.0002**	0.73	0.0002	0.21	0.0002	0.00028[g]
Chlordane	**2.4**	**0.0043**	**0.09**	**0.004**	2.4	0.0043	0.09	0.004	0.00081[g]
VOCs									
1,1,2,2-Tetrachloroethane	na	na	na	na	na	na	na	na	4[g]
1,1,2-Trichloroethane	na	na	na	na	na	na	na	na	16[g]
1,1-Dichloroethene (1,1-dichloroethylene)	na	na	na	na	na	na	na	na	7,100
1,2-Dichloroethane	na	na	na	na	na	na	na	na	37
1,2-Dichloropropane	na	na	na	na	na	na	na	na	15[g]
Acrolein	na	na	na	na	**3**	**3**	na	na	9
Acrylonitrile	na	na	na	na	na	na	na	na	0.25[g]
Benzene	na	na	na	na	na	na	na	na	51[g]
Bromodichloromethane (dichlorobromomethane)	na	na	na	na	na	na	na	na	17[g]
Bromoform	na	na	na	na	na	na	na	na	140[g]
Bromomethane (methyl bromide)	na	na	na	na	na	na	na	na	1,500
Carbon tetrachloride	na	na	na	na	na	na	na	na	1.6[g]
Chlorobenzene	na	na	na	na	na	na	na	na	1,600
Chloroform	na	na	na	na	na	na	na	na	470
Dibromochloromethane (chlorobromomethane)	na	na	na	na	na	na	na	na	13[g]
Dichloromethane (methylene chloride)	na	na	na	na	na	na	na	na	590[g]
Ethylbenzene	na	na	na	na	na	na	na	na	2,100
Tetrachloroethene	na	na	na	na	na	na	na	na	3.3[g]
Toluene	na	na	na	na	na	na	na	na	15,000

Table 4-19, cont. Aquatic life WQC and human health WQC

Chemical	State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
	Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Acute[e]	Chronic[f]	Organisms Only
trans-1,2-Dichloroethene	na	na	na	na	na	na	na	na	10,000
Trichloroethene (trichloroethylene)	na	na	na	na	na	na	na	na	30[g]
Vinyl chloride	na	na	na	na	na	na	na	na	2.4[g]
Dioxins and Furans									
2,3,7,8 TCDD	na	na	na	na	na	na	na	na	5.1E-09[g]

Note: Underlined values are hardness-dependent, and were calculated using a hardness value of 100 mg/L, which is the default assumption when site-specific hardness data are not available. Existing site-specific data or site-specific data that may be collected can be used to adjust values rather than using a default hardness value of 100 mg/L. Bolded criteria are the lower of the state and federal criteria (state criteria are bolded if the state and federal criteria are the same). The lower of the human health criteria (when multiple criteria are available) is also bolded.

[a] Standards are from WAC 173-201A-240. Available from: http://apps.leg.wa.gov/wac/default.aspx?cite=173-201A-240 (accessed on June 4, 2010).

[b] Standards are from EPA AWQC (except where noted). EPA AWQC available from: http://www.epa.gov/waterscience/criteria/wqctable/ (accessed on June 4, 2010).

[c] Aquatic life WQC are based on dissolved concentrations for metals (except mercury) and total concentrations for mercury and organic compounds.

[d] Human health WQC are based on dissolved concentrations for all chemicals.

[e] Acute WQC are 1-hr average concentrations not to be exceeded more than once every 3 years, with the exception of silver and pesticide concentrations, which are instantaneous concentrations not to be exceeded at any time, or the PCB concentration, which is a 24-hr average not to be exceeded at any time.

[f] Chronic WQC are 4-day average concentrations not to be exceeded more than once every 3 years, with the exception of pesticide and PCB concentrations, which are 24-hr average concentrations not to be exceeded at any time.

[g] Human health WQC are based on 1×10^{-6} excess cancer risk for carcinogenic chemicals.

[h] Criterion represents the inorganic fraction of arsenic.

[i] Standards are from 40 CFR 131.36 (NTR), as referenced in WAC 173-201A-240. Available from: http://ecfr.gpoaccess.gov/cgi/t/text/text-idx?c=ecfr&sid=879a68e0f8b500cb27fc2f8df4ec7f56&rgn=div5&view=text&node=40:21.0.1.1.18&idno=40 (accessed on June 4, 2010).

[j] Criteria based on the biotic ligand model. The acute and chronic biotic ligand model -based criteria for copper would be 2.3 and 1.5 µg/L, respectively, assuming DOC = 0.5 mg/L, pH = 7.5, hardness = 85 mg/L, and temperature of 20°C.

[k] The freshwater aquatic life WQC for pentachlorophenol are pH-dependent; a pH of 7.8 was assumed, which is the default assumption.

[l] Aldrin is metabolically converted to dieldrin. Therefore, the sum of aldrin and dieldrin concentrations is compared with the dieldrin criteria.

[m] Standards are for endosulfan.

AWQC – ambient water quality criteria
BEHP - Bis(2-ethylhexyl) phthalate
BHC – benzene hexachloride
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
MCL – maximum contaminant level
na – not available
nc – not calculated
NTR – National Toxics Rule
PAH – polycyclic aromatic hydrocarbon
PCB – polyvinyl chloride
SVOC – semivolatile organic compound
TCDD – tetrachlorodibenzo-*p*-dioxin
VOC – volatile organic compound
WAC – Washington Administrative Code
WQC – water quality criteria

Table 4-20. Summary of detected chemicals in LDW surface water, freshwater and marine aquatic life WQC, and human health WQC

	Detection Summary					State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
		Detection Frequency		Concentration Range (µg/L)		Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
Chemical	Number of Samples	Number of Detects	%	Detected	Non-Detected (RLs)	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Consumption of Organisms
Metals and Trace Elements														
Filtered														
Antimony	159	157	99	0.0095 – 0.116	0.01 – 0.01	nc	nc	nc	nc	nc	nc	nc	nc	640
Arsenic	168	168	100	0.175 – 1.46	na	190	360	**36**	**69**	**150**	**340**	36	69	*0.14*[g,h]
Cadmium	166	156	94	0.0072 – 0.0795	0.0067 – 0.0073	1.0	3.7	9.3	42	**0.25**	**2.0**	**8.8**	**40**	nc
Chromium (hexavalent)	148	148	100	0.1 – 0.576	na	**10**	**15**	**50**	**1,100**	11	16	50	1,100	nc
Copper	155	152	98	0.37 – 1.89	0.354 – 0.628	**11**	**17**	**3.1**	**4.8**	na[j]	na[j]	3.1	4.8	nc
Lead	160	160	100	0.0077 – 0.553	na	**2.5**	**65**	**8.1**	**210**	2.5	65	8.1	210	nc
Mercury	15	14	93	0.00013 – 0.00071	0.0001 – 0.0001	nc	nc	nc	nc	nc	nc	nc	nc	0.15[i]
Nickel	146	136	93	0.19 – 1.5	0.118 – 0.387	160	1,400	**8.2**	**74**	**52**	**470**	8.2	74	4,600
Selenium	154	1	1	0.16	0.12 – 0.16	**5**	**20**	**71**	**290**	5	nc	71	290	4,200
Thallium	168	78	46	0.005 – 0.011	0.0043 – 0.0053	nc	nc	nc	nc	nc	nc	nc	nc	0.47
Zinc	166	166	100	0.75 – 5.39	na	**100**	**110**	**81**	**90**	120	120	81	90	26,000
Unfiltered														
Mercury	29	15	52	0.00013 – 0.00071	0.0001	**0.012**	2.1	**0.025**	**1.8**	0.77	**1.4**	0.94	1.8	nc
PAHs[k]														
Acenaphthene	4	4	100	0.016 – 0.030	na	nc	nc	nc	nc	nc	nc	nc	nc	990
Anthracene	4	4	100	0.0014 – 0.0023	na	nc	nc	nc	nc	nc	nc	nc	nc	40,000
Benzo(a)anthracene	4	4	100	0.00017 – 0.00036	na	nc	nc	nc	nc	nc	nc	nc	nc	0.018[g]
Benzo(a)pyrene	4	4	100	0.000020 – 0.000033	na	nc	nc	nc	nc	nc	nc	nc	nc	0.018[g]
Benzo(b)fluoranthene	4	4	100	0.000081 – 0.00015	na	nc	nc	nc	nc	nc	nc	nc	nc	0.018[g]
Benzo(k)fluoranthene	4	4	100	0.000027 – 0.00012	na	nc	nc	nc	nc	nc	nc	nc	nc	0.018[g]
Chrysene	4	4	100	0.00017 – 0.00040	na	nc	nc	nc	nc	nc	nc	nc	nc	0.018[g]
Dibenzo(a,h)anthracene	4	4	100	0.0000027 – 0.0000052	na	nc	nc	nc	nc	nc	nc	nc	nc	0.018[g]
Fluoranthene	4	4	100	0.0063 – 0.020	na	nc	nc	nc	nc	nc	nc	nc	nc	140
Fluorene	4	4	100	0.0099 – 0.020	na	nc	nc	nc	nc	nc	nc	nc	nc	5,300
Indeno(1,2,3-cd)pyrene	4	4	100	0.0000093 – 0.000022	na	nc	nc	nc	nc	nc	nc	nc	nc	0.018[g]
Pyrene	4	4	100	0.0031 – 0.0070	na	nc	nc	nc	nc	nc	nc	nc	nc	4,000
Phthalates (unfiltered)														
BEHP	94	19	20	0.14 – 23.8	0.14 – 1.31	nc	nc	nc	nc	nc	nc	nc	nc	*2.2*[g]
Di-n-butyl phthalate	94	9	10	0.25 – 1.2	0.24 – 0.46	nc	nc	nc	nc	nc	nc	nc	nc	*4,500*
Other SVOCs (unfiltered)														
Phenol	94	1	1	2.01 – 2.01	0.94 – 1.1	nc	nc	nc	nc	nc	nc	nc	nc	*860,000*

	Detection Summary					State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
		Detection Frequency		Concentration Range (µg/L)		Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
Chemical	Number of Samples	Number of Detects	%	Detected	Non-Detected (RLs)	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Consumption of Organisms
Polychlorinated Biphenyls (unfiltered)														
Total PCBs	15	15	100	0.0001318 – 0.003211	na	**0.014**	**2**	**0.03**	**10**	0.014	nc	0.03	nc	*0.000064*[g]
Pesticides[k]														
4,4′-DDD	4	4	100	0.0000063 – 0.000013	na	nc	nc	nc	nc	nc	nc	nc	nc	0.00031[g]
4,4′-DDE	4	4	100	0.000000034 – 0.000065	na	nc	nc	nc	nc	nc	nc	nc	nc	0.00022[g]
DDT (and metabolites)	4	4	100	0.000035 – 0.000019	na	***0.001***[l]	***1.1***[l]	***0.001***[l]	***0.13***[l]	*0.001*[l]	*1.1*[l]	*0.001*[l]	*0.13*[l]	0.00022[g, l]
Aldrin	4	4	100	0.000003 –0.0000059	na	nc	nc	***0.0019***	***1.3***	nc	***3.0***	nc	*1.3*	0.000050[g]
Dieldrin	4	4	100	0.000020 – 0.000033	na	nc	nc	nc	nc	***0.056***	***0.24***	***0.0019***	***0.71***	0.000054[g]
Aldrin/dieldrin (sum)	4	4	100	0.0000037 – 0.000039	na	***0.0019***	***2.5***	***0.0019***	***0.71***	nc	nc	nc	nc	nc
Endrin	4	4	100	0.0000064 – 0.000014	na	***0.0023***	*0.18*	***0.0023***	***0.037***	*0.036*	***0.086***	*0.0023*	*0.037*	0.06
Chlordane	4	4	100	0.0000013 – 0.0000039	na	***0.0043***	***2.4***	***0.004***	***0.09***	*0.0043*	*2.4*	*0.004*	*0.09*	0.00081[g]

Note: Underlined values are hardness-dependent, and were calculated using a hardness value of 100 mg/L, which is the default assumption when site-specific hardness data are not available. A chemical is listed if it was detected in at least one LDW surface water sample and if it has a freshwater or marine aquatic life WQC or a human health WQC. WQC are italicized if the type of water data available does not match the type of data intended for comparison to a specific criterion (i.e., total vs. dissolved fraction or total arsenic vs. inorganic arsenic). Surface water samples were collected as single grab samples, except the four samples analyzed for PAHs and pesticides, which were collected using SPMDs. Bolded criteria are the lower of the state and federal criteria (state criteria are bolded if the state and federal criteria are the same).

[a] Standards are from WAC 173-201A-240. Available from: http://apps.leg.wa.gov/wac/default.aspx?cite=173-201A-240 (accessed on June 4, 2010).

[b] Standards are from EPA AWQC (except where noted). EPA AWQC available from: http://www.epa.gov/waterscience/criteria/wqctable/ (accessed on June 4, 2010).

[c] Aquatic life WQC are based on dissolved concentrations for metals (except mercury) and total concentrations for mercury and organic compounds. Data are presented for dissolved concentrations (filtered samples) for all metals (except mercury). Data are presented for total concentrations (unfiltered samples) for mercury and organic compounds.

[d] Human health WQC are based on dissolved concentrations for all chemicals. Data are presented for dissolved concentrations (filtered samples) when available. Data are presented for total concentrations (unfiltered samples) when dissolved data were not available.

[e] Chronic WQC are 4-day average concentrations not to be exceeded more than once every 3 years on average, with the exception of pesticide and PCB concentrations, which are 24-hr average concentrations not to be exceeded at any time.

[f] Acute WQC are 1-hr average concentrations not to be exceeded more than once every 3 years on average, with the exception of silver and pesticide concentrations, which are instantaneous concentrations not to be exceeded at any time, or the PCB concentration, which is a 24-hr average not to be exceeded at any time.

[g] Human health WQC are based on 1 x 10^{-6} excess cancer risk for carcinogenic chemicals.

[h] Criterion represents the inorganic fraction of arsenic. The water data represent total arsenic (i.e., the sum of the organic and inorganic arsenic species).

[i] Standards are from 40 CFR 131.36 (NTR), as referenced in WAC 173-201A-240. Available from: http://ecfr.gpoaccess.gov/cgi/t/text/text-idx?c=ecfr&sid=879a68e0f8b500cb27fc2f8df4ec7f56&rgn=div5&view=text&node=40:21.0.1.1.18&idno=40 (accessed on June 4, 2010).

[j] Criteria based on the biotic ligand model. The acute and chronic biotic ligand model -based criteria for copper would be 2.3 and 1.5 µg/L, respectively, assuming DOC = 0.5 mg/L, pH = 7.5, hardness = 85 mg/L, and temperature of 20°C.

[k] PAH and pesticide concentrations were estimated using SPMDs; these concentrations more closely represent the dissolved fraction than the total fraction of the chemical in the water column.

[l] Standards are based on criteria for 4,4′-DDT.

[m] Aldrin is metabolically converted to dieldrin. Therefore, the sum of aldrin and dieldrin concentrations is compared with the dieldrin criteria.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
CFR – Code of Federal Regulations
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LDW – Lower Duwamish Waterway
na – not applicable
nc – no criteria
NTR – National Toxics Rule
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RL – reporting limit
SPMD – semipermeable membrane device
SVOC – semivolatile organic compound
WQC –water quality criteria

Table 4-21. Summary of detected chemicals in LDW seep water, freshwater and marine aquatic life WQC, and human health WQC

	Detection Summary					State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
		Detection Frequency		Concentration Range (µg/L		Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
Chemical	Number of Samples	Number of Detects	%	Detected	Non-Detected (RLs)	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Consumption of Organisms
Metals and Trace Elements														
Filtered														
Arsenic	29	26	90	0.054 – 253	50	190	360	**36**	**69**	**150**	**340**	36	69	*0.14[g,h]*
Cadmium	29	16	55	0.009 – 0.508	2 – 10	1.0	3.7	9.3	42	**0.25**	**2.0**	**8.8**	**40**	nc
Chromium	29	2	7	8 – 9	1.51 – 20	**10**	**15**	**50**	**1,100**	11	16	50	1,100	nc
Copper	29	11	38	4 – 22.8	2 – 10	**11**	**17**	**3.1**	**4.8**	na[j]	na[j]	3.1	4.8	nc
Lead	29	16	55	0.036 – 3	1 – 20	**2.5**	**65**	**8.1**	**210**	2.5	65	8.1	210	nc
Mercury	26	16	62	0.00062 – 0.0153	0.1	nc	nc	nc	nc	nc	nc	nc	nc	0.15[i]
Nickel	26	12	46	0.84 – 5.25	0.04 – 50	160	1,400	**8.2**	**74**	**52**	**470**	8.2	74	4,600
Selenium	10	4	40	50 – 120	50 – 200	**5**	**20**	**71**	**290**	5	nc	71	290	4,200
Silver	29	14	48	0.012 – 0.112	0.015 – 20	nc	3.4	nc	**1.9**	nc	**3.2**	nc	1.9	nc
Thallium	10	2	20	50 – 70	50 – 200	nc	nc	nc	nc	nc	nc	nc	nc	0.47
Zinc	29	19	66	3.29 – 161	4 – 20	**100**	**110**	**81**	**90**	120	120	81	90	26,000
Unfiltered														
Mercury	34	17	50	0.00061 – 0.65	0.1	**0.012**	2.1	**0.025**	**1.8**	0.77	**1.4**	0.94	1.8	nc
Other SVOCs (filtered)[k]														
1,3-Dichlorobenzene	46	1	2	58	0.50 – 10	nc	nc	nc	nc	nc	nc	nc	nc	960
1,4-Dichlorobenzene	44	1	2	40.2	0.50 – 10	nc	nc	nc	nc	nc	nc	nc	nc	190
Polychlorinated Biphenyls														
Total PCBs (unfiltered)	34	8	24	0.020 – 8.9	0.017 – 1.0	**0.014**	**2**	**0.03**	**10**	0.014	nc	0.03	nc	nc
Total PCBs (filtered)[k]	16	1	6	0.26	0.017	nc	nc	nc	nc	nc	nc	nc	nc	0.000064[g]
Pesticides														
Heptachlor epoxide (unfiltered)	13	1	8	0.0076 – 0.0090	0.00080 – 0.016	nc	nc	nc	nc	**0.0038**	**0.52**	**0.0036**	**0.053**	nc
Heptachlor epoxide (filtered)[k]	16	1	6	0.0090	0.00080 – 0.016	nc	nc	nc	nc	nc	nc	nc	nc	0.000039[g]
VOCs (unfiltered)[l]														
1,1-Dichloroethene	96	5	5	1.0 – 27	1.0 – 20	nc	nc	nc	nc	nc	nc	nc	nc	*7,100*
1,2-Dichloroethane	96	6	6	1.0 – 27	1.0 – 20	nc	nc	nc	nc	nc	nc	nc	nc	*37*
1,2-Dichloropropane	96	3	3	1.0 - 16	1.0 - 20	nc	nc	nc	nc	nc	nc	nc	nc	*15[g]*
Benzene	96	5	5	2.2 – 36	1.0 – 10	nc	nc	nc	nc	nc	nc	nc	nc	*51[g]*
Chlorobenzene	96	3	3	4.1 – 8.8	1.0 – 20	nc	nc	nc	nc	nc	nc	nc	nc	*1,600*
Tetrachloroethene	96	24	25	1.0 – 760	1.0 – 20	nc	nc	nc	nc	nc	nc	nc	nc	*3.3[g]*

	Detection Summary					State WQC (µg/L)[a]				Federal AWQC (µg/L)[b]				
		Detection Frequency		Concentration Range (µg/L		Freshwater[c]		Marine[c]		Freshwater[c]		Marine[c]		Human Health[d]
Chemical	Number of Samples	Number of Detects	%	Detected	Non-Detected (RLs)	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Chronic[e]	Acute[f]	Consumption of Organisms
Toluene	96	2	2	2.0 – 2.7	1.0 – 20	nc	nc	nc	nc	nc	nc	nc	nc	*15,000*
Trichloroethene	96	24	25	1.0 – 480	1.0 – 20	nc	nc	nc	nc	nc	nc	nc	nc	*30*[g]
Vinyl chloride	96	18	19	1.0 – 3500	0.010 – 2.0	nc	nc	nc	nc	nc	nc	nc	nc	*2.4*[g]

Note: Underlined values are hardness-dependent, and were calculated using a hardness value of 100 mg/L, which is the default assumption when site-specific hardness data are not available. A chemical is listed if it was detected in at least one LDW seep water sample and if it has a freshwater or marine aquatic life WQC or a human health WQC. Seep water samples were collected as single samples at one point in time. WQC are italicized if the type of water data available does not match the type of data intended for comparison to a specific criterion (i.e., total vs. dissolved fraction or total arsenic vs. inorganic arsenic). Bolded criteria are the lower of the state and federal criteria (state criteria are bolded if the state and federal criteria are the same).

[a] Standards are from WAC 173-201A-240. Available from: http://apps.leg.wa.gov/wac/default.aspx?cite=173-201A-240 (accessed on June 4, 2010).

[b] Standards are from EPA AWQC (except where noted). EPA AWQC available from: http://www.epa.gov/waterscience/criteria/wqctable/ (accessed on June 4, 2010).

[c] Aquatic life WQC are based on dissolved concentrations for metals (except mercury) and total concentrations for mercury and organic compounds. Data are presented for dissolved concentrations (filtered samples) for all metals (except mercury). Data are presented for total concentrations (unfiltered samples) for mercury and organic compounds.

[d] Human health WQC are based on dissolved concentrations of the chemical for excess cancer risks of 1 x 10^{-6}, as presented in the NTR (40 CFR 131.36). Data are presented for dissolved concentrations (filtered samples) when available. Data for total concentrations are presented (unfiltered samples) when dissolved data were not available.

[e] Chronic WQC are 4-day average concentrations not to be exceeded more than once every 3 years on average, with the exception of pesticide and PCB concentrations, which are 24-hr average concentrations not to be exceeded at any time.

[f] Acute WQC are 1-hr average concentrations not to be exceeded more than once every 3 years on average, with the exception of silver and pesticide concentrations, which are instantaneous concentrations not to be exceeded at any time, or the PCB concentration, which is a 24-hr average not to be exceeded at any time

[g] Human health WQC are based on 1 x 10^{-6} excess cancer risk for carcinogenic chemicals.

[h] Criterion represents the inorganic fraction of arsenic. The water data represent total arsenic (i.e., the sum of the organic and inorganic arsenic species).

[i] Standards are from 40 CFR 131.36 (NTR), as referenced in WAC 173-201A-240. Available from: http://ecfr.gpoaccess.gov/cgi/t/text/text-idx?c=ecfr&sid=879a68e0f8b500cb27fc2f8df4ec7f56&rgn=div5&view=text&node=40:21.0.1.1.18&idno=40 (accessed on June 4, 2010).

[j] Criteria based on the biotic ligand model. The acute and chronic biotic ligand model -based criteria for copper would be 2.3 and 1.5 µg/L, respectively, assuming DOC = 0.5 mg/L, pH = 7.5, hardness = 85 mg/L, and temperature of 20°C.

[k] Data are presented for filtered samples. Data for the Boeing Plant 2 RFI, Terminal 117, and Rhône-Poulenc sampling events are from unfiltered samples, and it is not known if samples collected at Great Western from 1994 to 1999 were filtered. Data from those four areas were not included in this comparison to avoid combining results from both filtered and unfiltered samples.

[l] Data represent total concentrations because VOC samples were not filtered.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
CFR – Code of Federal Regulations
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LDW – Lower Duwamish Waterway
na – not applicable
nc – no criteria
NTR – National Toxics Rule
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RFI – Resource Conservation and Recovery Act facility investigation
RL – reporting limit
SVOC – semivolatile organic compound
VOC – volatile organic compound
WQC – water quality criteria

Table 4-22. Summary of detected chemicals in LDW porewater, freshwater and marine aquatic life WQC, and human health WQC

	DETECTION SUMMARY					STATE WQC (µg/L)[a]				FEDERAL AWQC (µg/L)[b]				
		DETECTION FREQUENCY		CONCENTRATION RANGE (µg/L)		FRESHWATER[c]		MARINE[c]		FRESHWATER[c]		MARINE[c]		HUMAN HEALTH[d]
CHEMICAL	NUMBER OF SAMPLES	NUMBER OF DETECTS	%	DETECTED	NON-DETECTED (RLs)	CHRONIC[e]	ACUTE[f]	CHRONIC[e]	ACUTE[f]	CHRONIC[e]	ACUTE[f]	CHRONIC[e]	ACUTE[f]	CONSUMPTION OF ORGANISMS
Metals and Trace Elements (unfiltered)														
Antimony	15	1	7	30	5 – 50	nc	nc	nc	nc	nc	nc	nc	nc	*640*
Arsenic	30	12	40	26 – 114	10 – 450	*190*	*360*	***36***	***69***	**150**	**340**	*36*	*69*	*0.14[g,h]*
Cadmium	30	4	13	4	5 – 30	*1.0*	*3.7*	*9.3*	*42*	**0.25**	**2.0**	**8.8**	**40**	nc
Copper	30	8	27	1 – 264	4 – 5	***11***	***17***	***3.1***	***4.8***	na[j]	na[j]	*3.1*	*4.8*	nc
Lead	30	13	43	0.6 – 4	1 – 250	***2.5***	***65***	***8.1***	***210***	*2.5*	*65*	*8.1*	*210*	nc
Mercury	30	10	33	0.0016 – 0.408	0.1 – 0.2	**0.012**	2.1	**0.025**	**1.8**	0.77	**1.4**	0.94	1.8	*0.15[i]*
Silver	15	6	40	0.3 – 0.5	1	nc	*3.4*	nc	***1.9***	nc	***3.2***	nc	*1.9*	nc
Zinc	30	6	20	4 – 1560	10 – 50	***100***	***110***	***81***	***90***	*120*	*120*	*81*	*90*	*26,000*
PAHs (unfiltered)														
Benzo(a)pyrene	9	1	11	0.032 – 0.032	0.37	nc	nc	nc	nc	nc	nc	nc	nc	*0.018[g]*
Benzo(b)fluoranthene	9	1	11	0.039 – 0.039	0.37	nc	nc	nc	nc	nc	nc	nc	nc	*0.018[g]*
Benzo(k)fluoranthene	9	1	11	0.039 – 0.039	0.37	nc	nc	nc	nc	nc	nc	nc	nc	*0.018[g]*
Fluoranthene	9	1	11	0.26 – 0.26	0.37	nc	nc	nc	nc	nc	nc	nc	nc	*140*
Pyrene	9	1	11	0.32 – 0.32	0.37	nc	nc	nc	nc	nc	nc	nc	nc	*4,000*
Phthalates (unfiltered)														
BEHP	9	5	56	2.0 – 390	1.9	nc	nc	nc	nc	nc	nc	nc	nc	*2.2[g]*
Other SVOCs (unfiltered)														
1,2-Dichlorobenzene	32	3	9	0.50 – 1.2	0.20 – 1.0	nc	nc	nc	nc	nc	nc	nc	nc	*1,300*
1,4-Dichlorobenzene	32	2	6	0.30	0.20 – 1.0	nc	nc	nc	nc	nc	nc	nc	nc	*190*
VOCs (unfiltered)														
1,1-Dichloroethene	32	3	9	0.30 – 4.9	0.20 – 1.0	nc	nc	nc	nc	nc	nc	nc	nc	*7,100*
1,2-Dichloroethane	32	2	6	7.4 – 15	0.20 – 1.0	nc	nc	nc	nc	nc	nc	nc	nc	*37*
1,2-Dichloropropane	32	2	6	1.7 – 2.5	0.20 – 1.0	nc	nc	nc	nc	nc	nc	nc	nc	*15[g]*
Benzene	41	1	2	9.4	0.20 – 4.7	nc	nc	nc	nc	nc	nc	nc	nc	*51[g]*
Chlorobenzene	32	4	12	0.30 – 1.4	0.20 – 1.0	nc	nc	nc	nc	nc	nc	nc	nc	*1,600*
Tetrachloroethene	32	2	6	0.40 – 1.1	0.20 – 1.0	nc	nc	nc	nc	nc	nc	nc	nc	*3.3[g]*
Toluene	41	5	12	0.30 – 3.5	0.20 – 2.0	nc	nc	nc	nc	nc	nc	nc	nc	*15,000*
trans-1,2-Dichloroethene	32	7	22	0.30 - 21	0.20 – 1.0	nc	nc	nc	nc	nc	nc	nc	nc	*10,000*
Trichloroethene	32	6	19	0.20 – 2.5	0.20 – 1.0	nc	nc	nc	nc	nc	nc	nc	nc	*30[g]*
Vinyl chloride	32	12	38	0.40 – 2,500	0.20 – 1.6	nc	nc	nc	nc	nc	nc	nc	nc	*2.4[g]*

Note: Underlined values are hardness-dependent, and were calculated using a hardness value of 100 mg/L, which is the default assumption when site-specific hardness data are not available. A chemical is listed if it was detected in at least one LDW porewater sample and if it has a freshwater or marine aquatic life WQC or a human health WQC. Porewater samples were collected as single samples at one point in time. WQC are italicized if the type of water data available does not match the type of data intended for comparison to a specific criterion (i.e., total vs. dissolved fraction or total arsenic vs. inorganic arsenic). Bolded criteria are the lower of the state and federal criteria (state criteria are bolded if the state and federal criteria are the same).

[a] Standards are from WAC 173-201A-240. Available from: http://apps.leg.wa.gov/wac/default.aspx?cite=173-201A-240 (accessed on June 4, 2010).

[b] Standards are from EPA AWQC (except where noted). EPA AWQC available from: http://www.epa.gov/waterscience/criteria/wqctable/ (accessed on June 4, 2010).

[c] Aquatic life WQC are based on dissolved concentrations for metals (except mercury) and total concentrations for mercury and organic compounds. Data are presented for total concentrations (unfiltered samples) for all chemicals because dissolved concentrations (filtered samples) were not available.

[d] Human health WQC are based on dissolved concentrations of the chemical for excess cancer risks of 1 x 10^{-6}, as presented in the NTR (40 CFR 131.36). Data are presented for total concentrations (unfiltered samples) because dissolved concentrations (filtered samples) were not available.

[e] Chronic WQC are 4-day average concentrations not to be exceeded more than once every 3 years on average, with the exception of pesticide and PCB concentrations, which are 24-hr average concentrations not to be exceeded at any time.

[f] Acute WQC are 1-hr average concentrations not to be exceeded more than once every 3 years on average, with the exception of silver and pesticide concentrations, which are instantaneous concentrations not to be exceeded at any time, or the PCB concentration, which is a 24-hr average not to be exceeded at any time.

[g] Human health WQC are based on 1 x 10^{-6} excess cancer risk for carcinogenic chemicals.

[h] Criterion represents the inorganic fraction of arsenic. The water data represent total arsenic (i.e., the sum of the organic and inorganic arsenic species).

[i] Standards are from CFR Title 40: Protection of Environment, Part 131 – Water Quality Standards, Subpart D: Federally Promulgated Water Quality Standards. Available from: http://ecfr.gpoaccess.gov/cgi/t/text/text-idx?c=ecfr&sid=879a68e0f8b500cb27fc2f8df4ec7f56&rgn=div5&view=text&node=40:21.0.1.1.18&idno=40 (accessed on June 4, 2010).

[j] Criteria based on the biotic ligand model. The acute and chronic biotic ligand model -based criteria for copper would be 2.3 and 1.5 µg/L, respectively, assuming DOC = 0.5 mg/L, pH = 7.5, hardness = 85 mg/L, and temperature of 20°C.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
LDW – Lower Duwamish Waterway
na – not applicable
nc – no criteria
NTR – National Toxics Rule
PAH – polycyclic aromatic hydrocarbon
RL – reporting limit
SVOC – semivolatile organic compound
VOC – volatile organic compound
WQC – water quality criteria

4.2.3 PCBs (Aroclors and congeners)

This section discusses the nature and extent of PCBs in surface sediment, subsurface sediment, tissue, surface water, and seep water. PCBs were not analyzed in porewater samples collected from the LDW.

4.2.3.1 Surface sediment

This section presents a general discussion of total PCB concentrations in surface sediment throughout the LDW based on data in the RI baseline dataset. In addition, surface sediment data are used to elucidate temporal patterns in total PCB concentrations, including: 1) 57 locations with sample pairs representing locations that were resampled at different points in time, 2) data from Slip 4, where multiple sediment investigations have been conducted in a fairly confined area, and 3) 7 locations surrounding the Duwamish/Diagonal EAA which have been monitored on an annual basis since 2003.

This section also discusses spatially weighted average concentrations (SWACs) of total PCBs (as Aroclors) in surface sediment, which were calculated using IDW. IDW methods have evolved throughout the project. As documented in a technical memorandum to the agencies (Windward 2006e), the IDW interpolation used for the ERA and FWM was based on optimization over broad areas of the LDW, considering a neighborhood of sample concentrations when determining each individual grid cell value. Thiessen polygons, rather than IDW, were used in the HHRA (Appendix B).

The interpolation methods described in the technical memorandum (Windward 2006e) required that the LDW be divided into three separate sections: north, central, and south. Because IDW will be used in the FS to a much greater extent than in the RI, the 2006 interpolation methods were re-evaluated for the FS with two additional objectives: 1) create a streamlined computational method that does not require division of the LDW into separate sections, and 2) optimize the interpolation parameters to enhance prediction accuracy (relative to the 2006 methods) at areas with moderate PCB concentrations. The resulting parameterization, which is used in this RI (except for the ERA and FWM, which use the 2006 interpolation) and will be used in the FS, can be implemented across the entire LDW in one step and is described in detail by ENSR (2008).

LDW-Wide Data

Surface sediment samples collected from 1,030 of the 1,365 locations in the RI baseline dataset were analyzed for total PCBs as Aroclors, and an additional 297 samples were analyzed for total PCBs using NOAA methods.[56] Total PCBs were detected at 94% of

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[56] These additional 297 samples were analyzed for selected PCB congeners, PCTs, and PCBs+PCTs using NOAA's HPLC/PDA and GC/ECD (equivalent to EPA 8082) methods. Total PCBs were calculated as the difference between total PCTs (which were determined using GC/ECD) and

the locations where PCB Aroclors were analyzed, with concentrations ranging from 1.6 to 220,000 µg/kg dw (Table 4-23). The highest total PCB concentrations were generally detected in EAAs (Maps 4-18 and 4-19), which are discussed in Appendix I. As presented in Table 4-23, the Aroclors detected most frequently in surface sediment were Aroclors 1254 and 1260, at 80 and 81% of the locations, respectively. Using the updated IDW interpolation method discussed above, the SWAC of total PCBs for the LDW from RM 0.0 to 5.25 was 350 µg/kg dw.

PCBs+PCTs, which were determined using HPLC/PDA. This method provides total PCB values but does not quantify the individual Aroclors.

Table 4-23. Summary of LDW-wide surface sediment data for detected Aroclors and total PCBs

Chemical	Detection Frequency		Unit	Concentration				SQS	CSL
	Ratio	%		Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]		
Aroclor 1242	108/1,021	11	µg/kg dw	7.8 J	2,700	49	0.87 – 6,100	na	na
Aroclor 1248	228/1,030	22	µg/kg dw	6.3	220,000	360	0.87 – 4,300	na	na
Aroclor 1254	817/1,022	80	µg/kg dw	2.2	110,000	500	1.3 – 4,300	na	na
Aroclor 1254/1260[c]	8/8	100	µg/kg dw	37	800	180	nc	na	na
Aroclor 1260	823/1,022	81	µg/kg dw	1.2 J	38,000	600	3.9 – 15,000	na	na
Aroclor 1262	2/12	17	µg/kg dw	270	840	97	3.3 – 20	na	na
Aroclor 1268	1/11	9	µg/kg dw	460 J	460 J	47	3.3 – 20	na	na
Total PCBs (dw)[d]	1,243/1,327	94	µg/kg dw	1.6 J	220,000	1,200	nc	na	na
Total PCBs (OC normalized)[e]	1,145/1,203	95	mg/kg OC	0.11 J	10,000	60	nc	12	65

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] RLs from non-detect samples only.

[c] Aroclors 1254 and 1260 could not be distinguished from each other in these eight samples, so they were reported together.

[d] Total PCBs (dw) represent the sum of the detected concentrations of the individual Aroclors. If no Aroclors were detected, the RL reported represents the highest RL for an individual Aroclor. As discussed in Section 4.1.3.3, 297 of these samples were analyzed using the NOAA method. which provides total PCB values but does not quantify the individual Aroclors.

[e] Summary statistics for OC-normalized PCB concentrations were calculated using only samples with TOC contents ≥ 0.5% and ≤ 4.0%. At very low or very high TOC contents, OC normalization is not appropriate for comparison to SMS (Michelsen and Bragdon-Cook 1993).

CSL – cleanup screening level
dw – dry weight
EPC – exposure point concentration
HHRA – human health risk assessment
J – estimated concentration
LDW – Lower Duwamish Waterway
na – not applicable
nc – not calculated
OC – organic carbon
PCB – polychlorinated biphenyl
RI – remedial investigation
RL – reporting limit
SQS – sediment quality standards
TEQ –toxic equivalent
TOC – total organic carbon

Area-based and numerical percentiles were calculated using the entire LDW dataset, which extends from RM 0.0 to RM 6.0. Five percent of the LDW area had a total PCB SWAC higher than 810 µg/kg dw (Table 4-24). The area-based percentiles derived from the IDW interpolations are lower than the numerical-based percentiles, as shown in Table 4-24, because many of the surface sediment samples were collected in more contaminated areas, and thus do not provide data that are representative of area-wide conditions.

Table 4-24. Percentiles of total PCB concentrations in LDW surface sediment collected from RM 0.0 to RM 6.0

	TOTAL PCB CONCENTRATION (µg/kg dw)[a]				
METHOD	ENTIRE DATASET	25TH PERCENTILE	50TH PERCENTILE	75TH PERCENTILE	95TH PERCENTILE
Area-based[b]	350[c] (SWAC)	48	110	210	810
Numerical	1,200[d] (mean)	58	140	390	4,300

[a] The UCL for total PCBs in surface sediment was 2,500 µg/kg dw using the RI baseline dataset, as calculated in Appendix C (Table C.3-8). This UCL is presented as the EPC for the netfishing scenario in the HHRA (Appendix B), which uses the entire surface sediment dataset.

[b] The 25th, 50th, 75th, and 95th percentiles calculated using the initial (Windward 2006e) IDW interpolation discussed in the ERA (Appendix A) and FWM were 51, 130, 220, and 1,117 µg/kg dw, respectively.

[c] The SWAC was calculated using surface sediment collected from RM 0.0 to RM 5.25.

[d] When calculating the mean concentration of total PCBs, if none of the individual Aroclors were detected in a given sample, then one-half the highest RL for an individual Aroclor was used for that sample.

dw – dry weight
EPC – exposure point concentration
HHRA – human health risk assessment
LDW – Lower Duwamish Waterway
PCB – polychlorinated biphenyl
RI – remedial investigation
RM – river mile
SWAC – spatially weighted average concentration
UCL – upper confidence limit on the mean

The maximum total PCB concentration was detected in 1995 at location NFK305, on the east side of the LDW between RM 4.8 and RM 4.9 (Map 4-18). Other areas with total PCB concentrations greater than the area-based 95th percentile concentration (810 µg/kg dw) were as follows: near RM 0.5 and RM 0.6 on the east side of the LDW (in the Duwamish/Diagonal EAA), near RM 1.0 in the navigation channel, between RM 2.2 and RM 2.3 on the west side (in EAA 2), in the Slip 4 EAA, between RM 3.2 and RM 3.7 on the east side of the LDW (in the Boeing Plant 2/Jorgensen Forge EAA), between RM 3.5 and RM 3.7 on the west side (in the T-117 EAA), and near RM 4.9 (in the Norfolk EAA) (Map 4-19). Patterns of total PCB concentrations in surface sediment are discussed in greater detail in Section 4.2.3.3 relative to subsurface data.

Total PCB concentrations expressed on an OC-normalized basis exceeded the SQS but not the CSL at 314 surface sediment locations (24%) and exceeded the CSL at 174 locations (13%) (Map 4-20; Table 4-16). Total PCB concentrations were less than either the SQS or CSL at 63% of the locations (Figure 4-1). Most of the total PCB concentrations that exceeded the CSL were in samples collected in or near EAAs between RM 0.0 and RM 1.0, between RM 3.2 and RM 4.0, and in Slip 4 (Map 4-20).

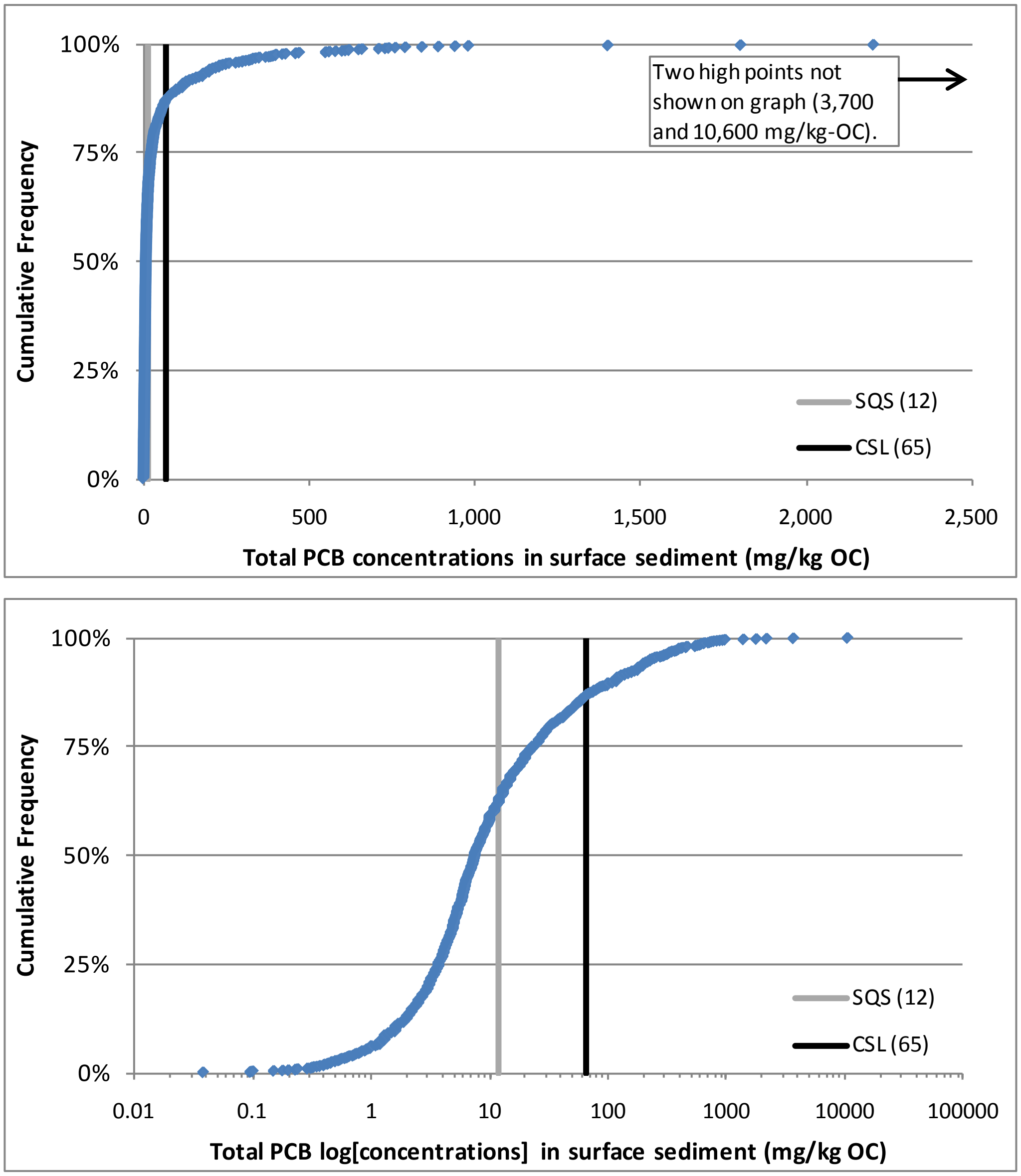

Figure 4-1. Cumulative frequency of OC-normalized total PCB concentrations in surface sediment (arithmetic and log-scale)

PCB congeners were analyzed in a subset of the surface sediment samples collected in 2005 for the RI and in samples collected during two sampling events in the late 1990s (the NOAA site characterization in 1997 and the EPA SI in 1998), as shown on Map 4-21. The PCB congener data collected in the late 1990s met DQOs, but RLs were relatively high. As part of the RI, surface sediment samples from 48 locations were analyzed for 26 PCB congeners.[57] The samples collected for PCB congener analyses

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[57] Discrete surface sediment grab samples were collected during Rounds 1 and 2 of the surface sediment sampling event at 32 locations, and composite sediment samples were co-located with benthic invertebrate tissue samples or clam tissue samples at eight locations each.

were selected to provide adequate spatial coverage of the LDW and to ensure that a range of locations with low, moderate, and high total PCB concentrations was represented. The PCB congeners analyzed included the 12 congeners identified by the World Health Organization (WHO) as having dioxin-like properties (i.e., PCB congeners 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, and 189). In addition, six principal PCB congeners (i.e., PCB congeners 66, 101, 110, 138, 153, and 180) were selected for analysis because they were detected in surface sediment at high concentrations relative to the other PCB congeners in the two historical datasets for the LDW. An additional eight PCB congeners that coelute with the PCB congeners listed above were also included in the analysis. In general, the PCB congeners with the highest concentrations at the locations sampled as part of the RI were PCB congeners 90, 110, 118, 129, and 153. Appendix E, Section E.6, presents summary statistics for all PCB congeners in surface sediment.

PCB TEQs were calculated using data for the 12 dioxin-like PCB congeners. Historical PCB congener data were not included in these TEQ calculations because of elevated RLs. Methods for mammalian PCB TEQ calculations are described in Appendix A, Section A.5.1.1. TEQ data for PCB congeners in surface sediment are presented in Table 4-25. Total PCB TEQs ranged from 0.0908 to 1,380 ng/kg dw, with a mean of 45.9 ng/kg. The dioxin-like PCB congener that contributed most to the total PCB TEQ in surface sediment was PCB-126. Based on a regression analysis, concentrations of total PCBs and total PCB TEQs in sediment were significantly correlated ($R^2 = 0.99$) for the 48 locations analyzed for PCB congeners in 2005.

Table 4-25. Sediment PCB TEQs in surface sediment samples

CHEMICAL	DETECTION FREQUENCY		TEQ (ng/kg dw)		
	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	CALCULATED MEAN[a]
Total PCB TEQ	48/48	100	0.0908 J	1,380	45.9
PCB-077 TEQ	48/48	100	0.00104	8.05	0.429
PCB-081 TEQ	48/48	100	0.000119	2.09	0.0810
PCB-105 TEQ	48/48	100	0.00184	110	3.01
PCB-114 TEQ	48/48	100	0.0000825	6.21	0.167
PCB-118 TEQ	48/48	100	0.00462	360	9.42
PCB-123 TEQ	48/48	100	0.0000837	4.14	0.116
PCB-126 TEQ	48/48	100	0.0758	798	29.1
PCB-156 TEQ	48/48	100	0.000825	53.7	1.43
PCB-167 TEQ	48/48	100	0.000318	15.5	0.427
PCB-169 TEQ	0/48	0	nd	nd	nc
PCB-189 TEQ	48/48	100	0.0000918	1.97	0.0715

Note: PCB TEQs were calculated using TEFs for mammals presented in Van den Berg et al. (2006). Total PCB TEQs were calculated for each sample by summing the TEQs for each dioxin-like congener. TEQs for individual congeners were calculated as the product of individual PCB congener concentrations and PCB congener-specific TEFs. If an individual PCB congener was not detected, the TEF for that congener was multiplied by one-half the RL for that congener.

[a] Calculated mean concentration is the mean of detected concentrations and one-half the RL for non-detected results.

PCB – polychlorinated biphenyl
TEF – toxic equivalency factor
TEQ – toxic equivalent
dw – dry weight
nc – not calculated
nd – not detected

LDW Resampled Locations

This section presents an evaluation of the change in total PCB concentrations in surface sediment between older samples and newer samples collected within 10 ft of each other.[58] Total PCBs had to be detected in at least one of the two samples to be included in the evaluation. Changes in concentrations could result from the mixing of surface sediment, burial of older sediment by newer sediment with different concentrations, transport of older sediment away from an area, or from resampling slightly different locations in areas with spatial heterogeneity in surface sediment chemical concentrations. Data from the Duwamish/Diagonal area and the Norfolk area were not included in this evaluation; monitoring data for these areas are discussed in detail later in this section for the Duwamish/Diagonal perimeter area, in Appendix I, Section I.4.1 (Duwamish/Diagonal cap), and in Appendix I, Section I.4.8 (Norfolk).

Of the 57 resampled surface sediment locations, total PCB concentrations decreased over time at 28 locations (49%), increased at 18 locations (32%), and were indistinguishable from the initial concentrations at 9 locations (16%)[59] (Table 4-26; Map 4-22). Two resampled locations were not included in Table 4-26 because total PCBs were not detected in either of the paired samples; RLs ranged from 19 to 40 µg/kg dw. At 26 of the 46 locations where detected total PCB concentrations increased or decreased, the concentration increase or decrease was ≤ 250 µg/kg dw.

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[58] The older data at these resampled locations were not included in the baseline dataset, as described in Section 4.1.2.1. The age difference between the older and newer samples ranged from 1 to 13 years, with an average difference of approximately 8 years.

[59] Concentration differences at these locations were within the range of analytical variability (≤ 25% increase or decrease compared to the initial concentration).

Table 4-26. Difference in total PCB concentrations at resampled locations presented by concentration difference

Approximate River Mile	Sampling Location ID		Sampling Date		Concentration (µg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
4.7	DR271	LDW-SS148	9/15/1998	3/9/2005	9,400	520	-8,880
3.6	SD-DUW90	SD-343-S	4/4/1996	8/27/2004	7,500	260 J	-7,240
3.5	WST323	T117-SE-10-G	10/21/1997	12/8/2003	7,900 J	1,200	-6,700
0.8	EST219	LDW-SS27	9/17/1997	1/18/2005	4,400 J	97 J	-4,303
1.4	DR030	LDW-SS50	8/17/1998	1/24/2005	4,800	590	-4,210
2.2	WIT280	B5a-2	10/3/1997	9/24/2004	5,200 J	1,730	-3,470
2.2	DR113	LDW-SS81	8/19/1998	3/8/2005	2,030 J	210	-1,820
3.9	EST144	LDW-SS123	9/25/1997	1/24/2005	1,500 J	149	-1,351
3.9	EIT061	LDW-SS121	9/29/1997	1/25/2005	2,400 J	1,060 J	-1,340
2.1	CH0023	LDW-SS79	10/16/1997	1/24/2005	1,200 J	68	-1,132
0.3	DUD042	LDW-SS17	11/11/1995	1/24/2005	1,060 J	120	-940
2.0	R7	LDW-SS75	10/15/1997	1/21/2005	1,200	520	-680
1.2	DR088	LDW-SS40	8/31/1998	1/18/2005	1,010 J	510 J	-500
3.9	R30	LDW-SS119	10/11/1997	1/19/2005	1,250 J	880 J	-370
0.2	DR035	LDW-SS12	8/11/1998	1/17/2005	516 J	171 J	-345
3.7	R21	LDW-SS113b	10/9/1997	1/20/2005	200	18 J	-182
4.2	R42	LDW-SS129	10/13/1997	1/20/2005	193	19 U	-174
1.3	DR053	LDW-SS44	8/31/1998	1/21/2005	260 J	103 J	-157
1.5	DR123	LDW-SS57	9/14/1998	1/24/2005	900	750	-150
2.1	DR106	LDW-SS76	8/19/1998	1/20/2005	227	117	-110
4.1	A11-05	LDW-SS126	8/18/1994	1/20/2005	109	20 U	-89
4.2	R45	LDW-SS130	10/16/1997	1/20/2005	101	26	-75
1.0	DR020	LDW-SS31	8/17/1998	1/21/2005	169 J	96	-73
4.2	R40	LDW-SS127	10/13/1997	1/20/2005	119	58	-61
0.2	K-05	LDW-SS10	9/27/1991	1/17/2005	91 J	31	-60
0.3	DR079	LDW-SS15	8/24/1998	1/17/2005	187 J	128 J	-59
2.8	DR175	LDW-SS94	8/20/1998	1/21/2005	120	72	-48
4.3	DR286	B10b	8/26/1998	8/19/2004	54	9.8 J	-44.2
1.0	DR019	LDW-SS32	8/17/1998	1/18/2005	162	122 J	**-40**
0	K-11	LDW-SS1	9/30/1991	1/17/2005	200 J	161 J	**-39**
1.7	DR097	LDW-SS63	8/20/1998	1/21/2005	126 J	95	**-31**
3.8	DR187	LDW-SS115	8/27/1998	1/25/2005	246	220	**-26**
3.2	DR202	LDW-SS104	8/27/1998	1/25/2005	98	75	**-23**
3.1	DR198	LDW-SS102	8/20/1998	1/24/2005	85	74	**-11**
1.9	DR131	LDW-SS70	8/13/1998	1/21/2005	97 J	96	**-1**

Approximate River Mile	Sampling Location ID		Sampling Date		Concentration (µg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
3.8	R24	LDW-SS117	10/10/1997	1/20/2005	73	79 J	**6**
1.4	DR065	LDW-SS52	8/17/1998	1/25/2005	185	209	**24**
4.2	EST135	B8b	11/12/1997	8/19/2004	10 J	37 J	27
2.4	WST342	DR141	10/23/1997	8/20/1998	38 J	68 J	30
1.4	DR160	LDW-SS51	8/12/1998	1/18/2005	115	155 J	40
1.0	WST367	DR048	9/19/1997	8/12/1998	29 J	88	59
0.1	K-07	LDW-SS4	9/30/1991	1/17/2005	87 J	153 J	66
4.2	DR242	SB-1	8/24/1998	8/25/2004	93 J	170	77
3.6	EST154	SD-334-S	9/24/1997	8/26/2004	150	290	140
3.6	SD-SWY07	SD-SWY17	6/13/1995	9/9/2003	320	460	140
1.4	DR028	B4b	8/17/1998	8/28/2004	207	400	193
0.9	DR021	LDW-SS319	8/17/1998	10/4/2006	142 J	350	208
2.6	EIT074	LDW-SS88	11/3/1997	1/25/2005	450	660	210
3.6	EST152	SD-309-S	9/24/1997	8/16/2004	290	570	280
0.9	DR085	LDW-SSB2b	8/31/1998	3/11/2005	413 J	790	377
2.8	EST180	LDW-SS92	10/6/1997	1/25/2005	230	970	740
3.7	DR186	LDW-SS111	8/27/1998	1/19/2005	1,180	3,200 J	2,020
3.7	SD-323-S	LDW-SS110	8/17/2004	1/25/2005	9,400 J	13,000	3,600
1.0	DR087	LDW-SS37	8/12/1998	1/18/2005	696	5,100	4,404
3.7	SD-DUW92	SD-320-S	4/2/1996	8/16/2004	1,500	8,900	7,400

Note: Concentrations in bold were within the range of analytical variability (≤ 25% increase or decrease compared with the initial concentration).

dw – dry weight
ID – identification
J estimated concentration

PCB – polychlorinated biphenyl
RL reporting limit
U not detected at RL shown

Three general areas had more than one location with decreases in total PCB concentrations of more than 250 µg/kg dw):

- **RM 0.2 to RM 0.4 –** At two locations in this area, concentrations decreased by 339 and 940 µg/kg dw; at two other resampled locations nearby, concentrations were relatively low and decreased by about 60 µg/kg dw at each location.
- **RM 2.0 to RM 2.3 –** Concentrations at four resampled locations in this area decreased by 680 to 3,470 µg/kg; the only other resampled location in this area (in Slip 3) had a decrease in concentration of 113 µg/kg dw.
- **RM 3.9 to RM 4.0 –** Concentrations at all the resampled locations in this area decreased by 370 to 1,351 µg/kg dw.

The remaining six locations with total PCB concentrations that decreased more than 250 µg/kg dw were at various single locations throughout the LDW.

Total PCB concentrations increased by more than 250 µg/kg dw at seven locations, most notably on the east side of the river near RM 3.7 (within the Boeing Plant 2/ Jorgensen Forge EAA), where there were three resampled locations with increases ranging from 2,020 to 7,400 µg/kg dw. There was also an area between RM 0.8 and RM 1.1 in the navigation channel with two resampled locations where PCB concentrations increased by 380 and 4,410 µg/kg dw.

The changes in total PCB concentrations over time at the resampled locations could reflect actual differences within an area from changes in sources or as a result of localized disturbances, or could reflect the heterogeneity of the sediment samples in a given location.[60] In addition, there is uncertainty in the accuracy of the recorded sampling locations as a result of inherent measurement error in the differential global positioning system (GPS) used in the sampling surveys, so the exact locations of two samples with coordinates within 10 ft of each other cannot be determined.[61] Therefore, although there is some uncertainty in the interpretation of these data, the more frequent observation of lower concentrations over time may indicate a decreasing trend in total PCB concentrations at some locations in the LDW.

To determine if total PCB concentrations were significantly different at locations that were resampled (i.e., if the mean difference between the paired concentrations was greater than zero), a non-parametric, paired t-test (Wilcoxon signed-rank test) was used. Because some of the differences were very large, a non-parametric test was considered more appropriate than a parametric test. The non-parametric test is more powerful than a parametric test when the data are not normally distributed and only slightly less powerful if data are normal. The test was run separately for each reach of the LDW (as defined in the STM) and for the entire LDW. Data for five locations were not included the analysis presented in Table 4-27 because these locations were sampled less than 6 years apart or where PCBs were not detected in either the initial or later samples.

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[60] Field duplicate samples (i.e., sample splits from a grab sample after homogenization) provide information on heterogeneity related to sample homogenization and laboratory variability. The relative percent difference in total PCB concentrations between paired field duplicate samples in the RI (n = 11) ranged from 1.6 to 190%, with an average of 39%.

[61] The differential GPS used for Phase 2 surface sediment sampling had a measurement error of approximately 3 to 6 ft. Prior to 2001, GPS technology was less accurate, so measurement errors may have been greater for samples collected before 2001.

Table 4-27. Results of statistical analyses of differences in total PCB concentrations in surface sediment at resampled locations

REACH OF THE LDW	n[a]	MEAN CONCENTRATION (µg/kg dw)		SIGNIFICANCE (p value)
		INITIAL	RESAMPLED	
RM 0.0 to RM 2.2	26	990	500	**0.027**
RM 2.2 to RM 4.0	18	1,400	1,100	0.74
RM 4.0 to RM 4.8	6	1,600	140	0.35
Entire LDW	50	1,200	660	**0.042**

[a] Data were not included for five locations (R42, A11-05, WST342, WST367, and SD-323-S) because they were sampled less than 6 years apart or PCBs were not detected in either the initial or later samples.

dw – dry weight

LDW – Lower Duwamish Waterway

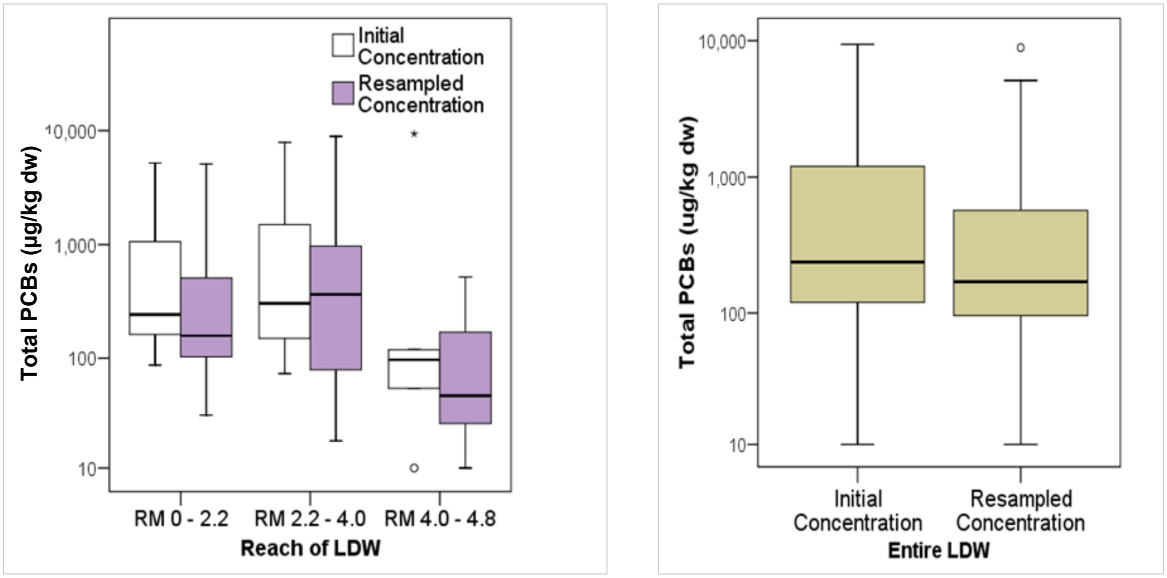
n – number of samples

RM – river mile

PCB – polychlorinated biphenyl

Bold identifies p values that are significant at the 95% confidence level.

When the data were analyzed by reach of the LDW, the differences between initial and resampled total PCB concentrations were significant at the 95% confidence level in the reach between RM 0.0 and RM 2.2 (Table 4-27; Figure 4-2). Differences in the other reaches were not significant at the 95% confidence level. When data were analyzed for the entire LDW, differences between the initial and resampled total PCB concentrations were significant at the 95% level.



Note: The boxes indicate the 25th and 75th percentiles within which 50% of the data lie. The length of this box indicates the interquartile range (IQR). The line in the center of the box is the 50th percentile (median). The whiskers of the box plot indicate the maximum and minimum values from the dataset that are within 1.5 IQRs of the 25th and 75th percentile values. Outliers are indicated by a circle and have a value between 1.5 and 3 of the IQR from the upper or lower edge of the box (75th or 25th percentile). Extreme values are indicated by an asterisk and have a value > 3 times the IQR from the 75th or 25th percentile. Data were not included for three locations (WST342, WST367, and SD-323-S) that were sampled less than 1year apart (all other paired locations were sampled at least 6 years apart). Therefore, the maximum value of 13,000 µg/kg dw from location SD-323-S was not included in this figure. Also, data were not included for two locations (R42 and A11-05) where PCBs were not detected in the later sample.

Figure 4-2. Differences in total PCB concentrations in surface sediment at resampled locations

Slip 4

Slip 4, located near RM 2.8 on the east side of the LDW (Map 4-23), is one of seven areas within the LDW that have been identified as an EAA[62] (Windward 2003e). Surface sediment data for PCBs have been collected during six sampling events conducted in Slip 4 between 1990 and 2004. Slip 4 is the only area in the LDW sampled during more than two time periods at a relatively high sampling density, providing an opportunity to evaluate the data for temporal trends on an area-specific basis. The western portion of Slip 4 was most recently dredged in 1996 to allow barge and tug access to the Crowley Marine Services pier; this dredged area is shown on Map 4-23.

The six surface sediment investigations conducted in Slip 4 included:

- Environmental site assessment in 1990 (Landau 1990) (9 surface sediment samples)
- NOAA sediment characterization study in 1997 (NOAA 1998) (18 surface sediment samples)

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[62] EAAs are discussed in greater detail in Section 9.

- Phase 1 site characterization study in 1997 (Exponent 1998) (7 surface sediment samples)
- EPA site inspection in 1998 (Weston 1999a) (7 surface sediment samples)
- EAA investigation in 2004 (Integral 2004) (30 surface sediment samples)
- RI benthic invertebrate sampling event in 2004 (Windward 2005a) (3 surface sediment samples collected with co-located clam samples)

To evaluate changes in total PCB concentrations over time, the data were classified by collection date into three temporal groups: 1990 (n = 9), 1997-1998 (n = 32), and 2004 (n = 33). Data were also classified spatially into four areas: two areas inside the EAA (head of the slip and mid-slip) and two areas outside the EAA (mouth of the slip and outside the slip). Map 4-23 shows the total PCB concentrations detected at each location during each time period, the boundary of the four areas, and the boundary of the currently approved Slip 4 EAA. No locations were sampled more than once at the same coordinates (i.e., within 10 ft of each other).

In all years, the highest concentrations were reported within the EAA boundary, and there was little overlap between concentrations within the EAA and outside the EAA (Map 4-23; Figures 4-3 and 4-4). Two-way analysis of variance and *post hoc* multiple comparison tests were conducted to evaluate the effects of location (inside vs. outside the EAA) and time (1990, 1997-1998, and 2004) on log-transformed PCB concentrations (in µg/kg dw). These tests indicated the following:

- The difference in mean log total PCB concentrations between 1990 and 1997-1998 was not significant either within or outside of the EAA at Slip 4.[63]
- The difference in mean log total PCB concentration between 1997-1998 and 2004 means was significant in both areas.[64]
- The difference in mean log total PCB concentrations between 1990 and 2004 was not significant in the area outside the EAA but was significant within the EAA.[65]

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[63] Year effect was significant ($p < 0.0005$) in two-way ANOVA testing for effects of year, area, and area-by-year interaction. P values for *post hoc* one-way ANOVA testing for effects of year within area, using Bonferonni correction for multiple comparisons, were as follows: $p = 0.252$ within EAA, $p = 1.000$ outside EAA, indicating non-significant differences.

[64] P values for *post hoc* one-way ANOVA testing for effects of year within area using Bonferonni correction for multiple comparisons, were as follows: $p < 0.0005$ within EAA, $p = 0.005$ outside EAA.

[65] P values for *post hoc* one-way ANOVA testing for effects of year within area using Bonferonni correction for multiple comparisons, were as follows: $p < 0.0005$ within EAA, $p = 1.000$ outside EAA.

Uncertainties associated with this analysis include the low number of samples and minimal spatial coverage in the slip in the 1990 sampling, and the fact that the same locations were not re-sampled in the three different time periods.

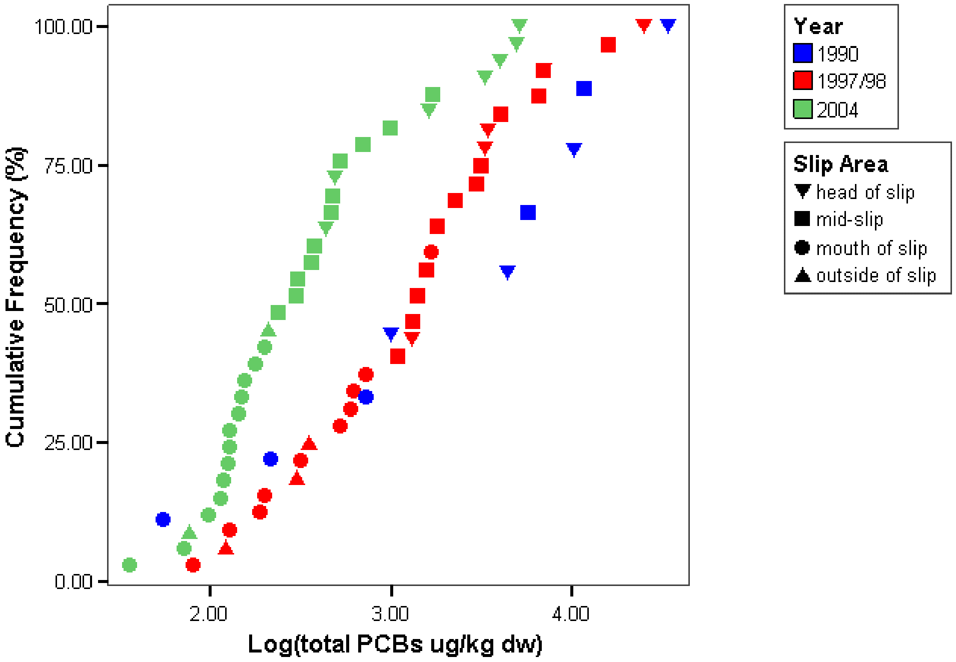
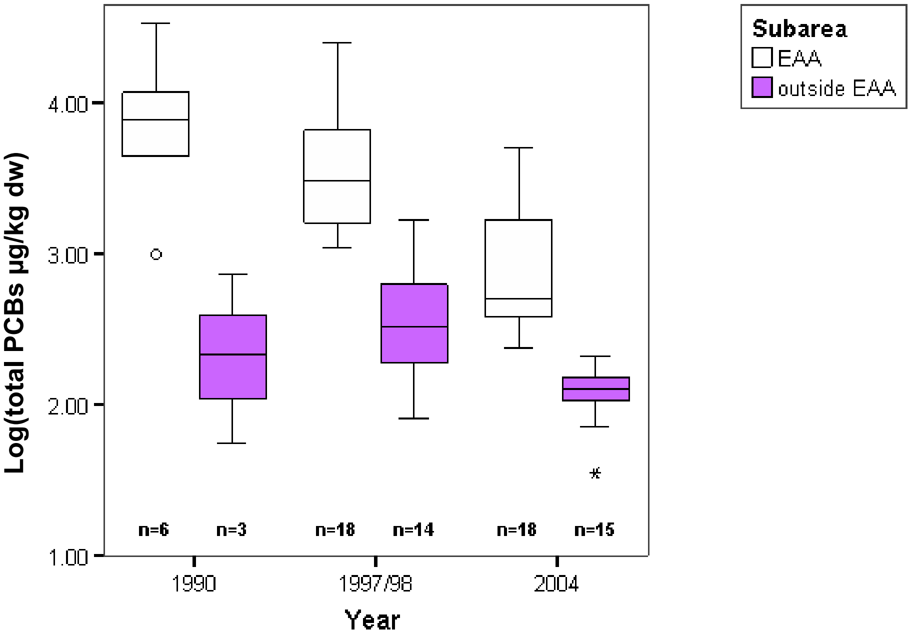


Figure 4-3. Cumulative frequency distributions of total PCB concentrations in Slip 4 surface sediment



Note: The boxes indicate the 25th and 75th percentiles within which 50% of the data lie. The length of this box indicates the interquartile range (IQR). The line in the center of the box is the 50th percentile (median). The whiskers of the box plot indicate the maximum and minimum values from the dataset that are within 1.5 IQRs of the 25th and 75th percentile values. Outliers are indicated by a circle and have a value between 1.5 and 3 of the IQR from the upper or lower edge of the box (75th or 25th percentile). Extreme values are indicated by an asterisk and have a value > 3 times the IQR from the 75th or 25th percentile.

Figure 4-4. Boxplot of total PCB concentrations in Slip 4 sediment

Duwamish/Diagonal Perimeter

Surface sediment sampling has been conducted at the Duwamish/Diagonal EAA on an annual basis since 2003 to monitor sediment quality following remedial actions. This section discusses total PCB data collected at seven locations along the perimeter of the remediated area that have been monitored during five sampling events (Map 4-24). Any samples collected within 200 ft after completion of dredging and capping of the remediated area in 2004 were excluded from the RI baseline dataset. The baseline dataset represents pre-dredge conditions, and these sediments could have been influenced by dredge residuals.[66] Although not in the RI baseline dataset, these perimeter data are presented in this section and in Appendix I, Section I.4.1 to evaluate changes at these locations over time. Monitoring data for samples collected within the capped area are also discussed in Appendix I, Section I.4.1.

Remediation at the Duwamish/Diagonal EAA consisted of dredging sediment from the area between RM 0.4 and RM 0.6 from November 2003 to January 2004 (King County et al. 2005b). The dredged area was then capped between January and March

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[66] Dredge residuals affected the area south and west of Cap Area B and resulted in the need for placement of the thin-layer cap.

of 2004 by placing clean cap material over a 5-ac rectangle (Area A) and a 2-ac rectangle (Area B), as shown on Map 4-24. A 9-in. thin-layer placement was later placed over an area to the south and west of Area B in February 2005. Monitoring of surface sediment chemistry at the seven perimeter locations began in October 2003 before remediation began and was then conducted on an annual basis beginning in March 2004.

In general, total PCB concentrations in surface sediment at the seven perimeter monitoring locations showed a decreasing trend from 2004 to 2007 (Map 4-24), although trends were more variable at locations 1C and 11C.[67] At locations 1C, 2C, 8C, and especially 11C, it appears that lower concentrations in 2004 were likely affected by the higher sand content deposited over some of the perimeter areas during capping operations, as shown on Map 4-24. At most perimeter stations, the percentage of sand decreased in 2007, indicating the deposition of finer-grained material. As discussed in more detail in Appendix I, Section I.4. 1, monitoring data from capped Areas A and B showed different patterns. In Area A, total PCB concentrations were generally lowest in 2004 immediately after capping, higher in 2005 and 2006, and then lower again in 2007. In Area B, total PCB concentrations were relatively unchanged from 2004 through 2007.

4.2.3.2 Subsurface sediment

Subsurface sediment cores were sampled at various intervals during different sampling events. To calculate summary statistics, two categories were created according to specified intervals. The following intervals were selected: 2-ft intervals of 0 to 2 ft, 2 to 4 ft, 4 to 6 ft, 6 to 8 ft, and 8 to 10 ft; and 1-ft intervals (e.g., 0 to 1 ft, 1 to 2 ft) to a depth of 10 ft below mudline. Data were assigned to these intervals by first rounding (to the nearest foot) the top and bottom depths of each sample (in feet below mudline) and then assigning the data to the appropriate sampling interval. If data were available from two 1-ft intervals at a particular location, the concentrations were also averaged to obtain a calculated concentration for the corresponding 2-ft interval category; these intervals are presented in addition to the 1-ft intervals. For example, if a location had data in both the 0-to-1-ft and 1-to-2-ft intervals, the concentrations were averaged for a calculated concentration in the 0-to-2-ft interval.

Total PCBs were detected in 74% of the subsurface sediment samples, at concentrations ranging from 0.52 to 890,000 µg/kg dw (Table 4-28). Most of the subsurface samples analyzed for total PCBs were from the top 4 ft of the sediment cores. Data shown on Maps 4-25a through 4-25c and presented in Table 4-28 are discussed in more detail in Section 4.2.3.3.

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[67] The 2003 samples are not included in this analysis because they represent conditions before dredging.

Table 4-28. Summary of LDW-wide subsurface sediment data for total PCBs (as Aroclors)

SAMPLING INTERVAL (ft)	DETECTION FREQUENCY		TOTAL PCB CONCENTRATION (µg/kg dw)[a]	
	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT
All Data				
Any interval	609/821	74	0.52 J	890,000
1-ft Intervals[b]				
0 to 1	114/118	97	13.5	108,000
1 to 2	83/101	82	19.6	6,500
2 to 3	75/104	72	4 J	32,000
3 to 4	36/49	73	0.52 J	9,100
4 to 5	26/57	46	23	6,400
5 to 6	13/18	72	22	4,700
6 to 7	6/21	29	21	2,400
7 to 8	2/5	40	540	910
8 to 9	3/10	30	25	690
2-ft Intervals[b]				
0 to 2	130/137	95	12.9	890,000
2 to 4	106/135	79	3 J	19,000 J
4 to 6	47/60	78	3.9 J	15,000
6 to 8	20/30	67	4.5 J	3,800
8 to 10	12/24	50	14 J	540

Note: Data included in the subsurface dataset are described in Section 4.1.2.2; subsurface sediment samples collected from areas that were subsequently dredged are not summarized in this table because they do not represent existing conditions.

[a] Mean concentrations were not calculated because sampling was generally focused in contaminated areas, resulting in a non-random sampling pattern, which is not likely representative of the entire LDW.

[b] For the calculation of summary statistics, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to either the 1- or 2-ft sampling category, whichever best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were averaged to obtain a calculated concentration for the corresponding 2-ft interval. Some of the subsurface data were collected from intervals other than 1 or 2 ft; therefore, the detection frequency ratio for "any interval" may reflect data for intervals not included in these categories.

dw – dry weight

PCB – polychlorinated biphenyl

J – estimated concentration

Of the subsurface sediment samples collected and analyzed as part of the RI, 27 of the 47 samples analyzed for total PCBs were collected entirely within the lower alluvium unit. The lower alluvium unit is predominantly dense sand that was deposited prior to the industrialization of the area. The depth from the river bottom to the surface of this unit ranged widely from 0.2 to 12 ft, with an average depth of 6.2 ft, and is discussed in greater detail in Section 2.5.4.

Total PCB concentrations in subsurface sediment samples collected from entirely within the lower alluvium unit were ≤ 53 µg/kg dw and less than the SQS in all samples.[68]

4.2.3.3 Surface and subsurface sediment patterns

This section presents the spatial and vertical patterns of total PCB concentrations in surface and subsurface sediment by river mile. These patterns were assessed to identify areas with relatively high concentrations in surface and subsurface sediment, to evaluate the spatial relationships between the two, and to identify areas where maximum total PCB concentrations were detected in uppermost intervals or in deeper subsurface intervals. To limit the discussion, specific surface sediment locations were noted in the text if: 1) total PCB concentrations were higher in surface sediment samples than in sampling intervals of co-located subsurface cores, 2) the total PCB concentration in surface sediment was greater than the 75th percentile, 3) the total PCB concentration in surface sediment was at least 25% greater than the highest total PCB concentration in any subsurface interval, and 4) the location was outside of the Boeing Plant 2/Jorgensen Forge or T-117 EAAs (these areas are discussed in more general terms). Total PCB concentrations in all co-located surface and subsurface sediment samples are presented on Maps 4-26a through 4-26h. Maps 4-27a and 4-27b show only the subsurface sediment locations with co-located surface sediment data. The integration of this information with the physical CSM (as presented in Section 3) is discussed in Section 4.3.

The LDW was divided into five sections for discussion purposes (RM 0.0 to RM 1.0, RM 1.0 to RM 2.2, RM 2.2 to RM 3.0, RM 3.0 to RM 4.0, RM 4.0 to RM 5.0). Boundaries for some of these sections correspond to the river reaches defined by the STM (i.e., Reach 1 from RM 0.0 to RM 2.2, Reach 2 from RM 2.2 to RM 4.0, and Reach 3 from RM 4.0 to RM 4.8). Maps in this section present data separately for locations within the navigation channel, west of the channel, and east of the channel to facilitate comparisons of river bench vs. navigation channel areas and for consistency with results of sediment transport modeling (Maps 4-26a through 4-26 h). According to the

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[68] Subsurface data are compared to SMS criteria in Appendix J. One sample collected from the 6-to-8.8-ft interval at LDW-SC10 had a total PCB concentration exceeding the SQS of 12 mg/kg OC (35 mg/kg OC; 350 µg/kg dw). This sampling interval was identified as lower alluvium in the draft core logs presented in the subsurface sampling data report (Windward and RETEC 2007), but this sample contained silt and sand from the upper alluvium unit.

STM, bench and channel areas have some distinct differences in sedimentation rates and potential scour depths in some sections of the LDW.

It is important to note that the certainty and scale of the analyses presented in this section (and in Sections 4.2.4 and 4.2.5 for arsenic and cPAHs, respectively), are dictated by the availability of surface and subsurface sediment data. Sources of uncertainty include:

- Uncertainty in the interpretation of subsurface data dictated by the size of the sampling interval within each core, which is variable across the site.
- Uncertainty in distinguishing vertical patterns in locations with larger sampling intervals.
- Uncertainty in comparing surface sediment data to subsurface core data because samples may have been collected at different times and conditions at the surface may have changed; interpolated surface sediment data are particularly uncertain.

RM 0.0 to RM 1.0

Total PCB concentrations in surface and subsurface sediment samples collected between RM 0.0 and RM 1.0 were highly variable (Map 4-26a). Within this section of the LDW, three areas had total PCB concentrations in surface sediment greater than the area-based 95th percentile (810 µg/kg dw): 1) RM 0.1 on the eastern shoreline, 2) the vicinity of the Duwamish/Diagonal EAA (between RM 0.4 and RM 0.6), and 3) RM 0.6 on the western shoreline.

At RM 0.1 to RM 0.2 on the eastern shoreline, the peak total PCB concentrations in subsurface cores were detected in the 2-to-4-ft interval (2,900 µg/kg dw at LDW-SC2 and 600 µg/kg dw at LDW-SC4). In a finer-resolution core collected just south of Harbor Island near an SD (LDW-SC1 sampled at 0.5-ft intervals), there was a peak in total PCB concentrations (6,700 µg/kg dw) in the 1-to-1.5-ft interval. Just east of the navigation channel at RM 0.15 (LDW-SC3), PCBs were not detected in subsurface sediment samples.

On the west side, a core (LDW-SC5) was collected near a small area at RM 0.15; the peak total PCB concentration was detected in the uppermost 1-ft interval (510 µg/kg dw) and concentrations decreased with depth. Farther south, at RM 0.3, the peak total PCB concentration was detected in the 4-to-4.5-ft interval (2,600 µg/kg dw at LDW-SC6). A historical core collected 290 ft to the south (DR068) had the same concentration (2,600 µg/kg dw) in the only interval sampled (0 to 2 ft).

A relatively large number of cores from within and near the Duwamish/Diagonal EAA have been analyzed for total PCBs. Sediments within the EAA were either dredged and capped or covered with a thin layer of sand subsequent to the collection and analysis of these samples. Prior to dredging and capping, total PCB concentrations in this area were high in surface sediment (greater than the 75th percentile) and in subsurface sediment (generally having at least one interval greater than the 95th percentile). A cluster of cores near the center of the dredged area at RM 0.5 (DUD254, DUD253, DUD020, DUD006, DR008, DUD256, and LDW-SC10) had peak PCB concentrations below 2 ft. In three other cores from the dredged area (DUD252, DUD251, and DUD255) and cores from the thin-layer cap area (DUD257, DUD260, DUD262, DUD027, and DUD261), peak total PCB concentrations were in the uppermost intervals. In the co-located surface sediment sample at the DUD027 subsurface core location, the total PCB concentration (56,200 µg/kg dw) was higher than the peak concentration in the subsurface core. At another sampling location just west of the EAA (LDW-SC9), the total PCB concentration (4,610 µg/kg dw) in the co-located surface sediment sample was also higher than the peak concentration in the subsurface core. There is high uncertainty associated with the interpretation of data from cores collected within the Duwamish/Diagonal EAA because many of them were collected at 3-ft intervals.

Relatively few subsurface sediment core samples have been collected from the Kellogg Island area. Interpolated total PCB concentrations in surface sediment throughout this area were generally low (≤ 110 µg/kg dw), with a few higher concentrations (between the 75th and 95th percentiles) on the west side of the island. Two cores collected at the same location north of the island at RM 0.6 had peak PCB concentrations (~2,000 µg/kg dw) at depth (in the 2-to-2.5-ft interval at LDW-SC12 and in the 2-to-4-ft interval at DR044).

On the western shoreline of the LDW to the north of Kellogg Island, a single core (LDW-SC11) was collected, with high total PCB concentrations (3,000 µg/kg dw) in the uppermost interval (0 to 0.8 ft);PCBs were not detected below this interval. In a core collected south of the island (LDW-SC19) in an area of low interpolated PCB concentrations in surface sediment, PCBs were highest in the 6-to-7-ft interval (2,400 µg/kg dw).

Interpolated total PCB concentrations were generally between the 50th and 95th percentile in surface sediments in Slip 1. At the head of the slip, total PCB concentrations in core LDW-SC17 were highest at depth (9,800 µg/kg dw in the 2-to-4-ft interval), although all intervals had high concentrations (greater than the 95th percentile). Near the mouth of the slip, peak total PCB concentrations (ranging from 1,950 to 5,400 µg/kg dw) were detected at depth (in the 2-to-4-ft and 4-to-6-ft intervals) in all three cores collected in this area (LDW-SC15, LDW-SC16, and DR021).

Two cores were collected just downstream of Slip 1 at RM 0.85. In the navigation channel, the core from LDW-SC14 had high total PCB concentrations (1,550 to 4,500 µg/kg dw) in the uppermost intervals down to 4.1 ft, with the peak concentration in the 0-to-1.4-ft interval. Near the eastern shoreline of the LDW (LDW-SC13) the highest total PCB concentrations were in the uppermost intervals down to 2 ft, but concentrations were somewhat lower, ranging from 280 to 470 µg/kg dw.

In summary, the LDW between RM 0.0 and RM 1.0 is highly variable. Interpolated concentrations in surface sediment covered all percentile categories, although only a few areas had concentrations greater than the 95th percentile. Many of the areas sampled with subsurface cores had peak total PCB concentrations at depth. Exceptions to this general pattern include a few locations along the western shoreline (LDW-SC5 and LDW-SC11), a location in the navigation channel near RM 0.9, and a number of locations in the Duwamish/Diagonal EAA where peak concentrations were detected in the uppermost interval of the cores.

RM 1.0 to RM 2.2

Total PCB concentrations in surface and subsurface sediment samples collected between RM 1.0 and RM 2.2 were also variable (Map 4-26b). One area just south of RM 1.0 in the navigation channel had the peak total PCB concentration in the uppermost interval (3,200 µg/kg dw in LDW-SC20 in the 0-to-2-ft interval). In the co-located surface sediment sample associated with that core, the total PCB concentration was higher, at 5,100 µg/kg dw. In the core collected on the western shoreline at RM 1.0 (LDW-SC21), the peak total PCB concentration (1,680 µg/kg dw) was detected at depth (4 to 6.2 ft). Cores along the eastern shoreline at RM 1.0 (C2, C3, A1, LDW-SC18, and LDW-SC22) had relatively lower PCB concentrations, but samples from within the historical cores were collected over relatively large intervals (e.g., 4-ft intervals); thus, it was not possible to determine the depth or magnitude of peak concentrations. At sampling location LDW-SC18, the total PCB concentration (650 µg/kg dw) in the co-located surface sediment sample was higher than the peak concentration in the subsurface core.

Total PCB concentrations in surface sediment areas along the eastern shoreline between RM 1.2 and RM 1.4 were variable. Peak concentrations in cores in this area were detected at depth, with peak concentrations in the 4-to-6-ft, 2-to-4-ft, and 1-to-1.5-ft intervals in cores LDW-SC23, DR025, and LDW-SC27, respectively.

Just downstream of Slip 2 along the eastern shoreline, numerous historical cores and two RI cores were collected and analyzed for PCBs. The historical cores were collected prior to dredging in this area and were generally composited over larger intervals (0 to 3 ft or 0 to 4 ft), so the depth of the peak total PCB concentration is unknown. In the cores collected as part of the RI, peak concentrations (12.9 µg/kg dw at LDW-SC30 and 370 µg/kg dw at LDW-SC31) were in the uppermost intervals.

One core (LDW-SC32) was located within Slip 2. Total PCB concentrations in the uppermost 4 ft of this core were higher than the concentration in the co-located surface sediment sample. The peak total PCB concentration of 2,450 µg/kg dw in subsurface sediment was detected in the 2-to-4-ft interval, compared with the concentration of 211 µg/kg dw in surface sediment.

At the head of Slip 3, a subsurface peak of 950 µg/kg dw was detected in the 1-to-2-ft interval of core LDW-SC37. Near the mouth of Slip 3, in cores DR106 and LDW-SC36, subsurface concentrations of PCBs were low (less than the 50th percentile). In the core just upstream from Slip 3 (DR112), the peak concentration was detected in the 2-to-4-ft interval (330 µg/kg dw). Thus, peak concentrations were detected at depth at the head and just downstream of the slip, and were relatively low towards the mouth of the slip.

Multiple historical cores from the navigation channel between RM 1.7 and RM 2.0 were analyzed for total PCBs. These cores, collected over intervals from 0 to 4 ft, had variable total PCB concentrations, ranging from below the 25th percentile to the 95th percentile. Vertical patterns were not evident based on these cores because of the large sampling intervals. Core LDW-SC33, just east of the navigation channel at RM 1.9, was analyzed at 0.5-ft intervals down to 3 ft and had a peak total PCB concentration of 4,700 µg/kg dw in the 1-to-1.5-ft interval.[69]

Six locations between RM 1.2 and RM 1.5 on the west side of the navigation channel had sufficient data to determine the depth of peak concentrations within these cores. Two of these cores had the highest concentrations in the uppermost intervals (33 and 280 µg/kg dw in LDW-SC29 and LDW-SC24, respectively). Cores LDW-SC26 and LDW-SC28 had peak concentrations of 2,300 and 3,200 µg/kg dw in their 6-to-8-ft intervals. Peak concentrations were somewhat lower (750 to 800 µg/kg dw) and shallower (2 to 4 ft or 4 to 6 ft) in the other two cores (LDW-SC25 and DR054). Thus, in this area, peak concentrations were detected at depth in the cores with the highest subsurface concentrations.

Along the western shoreline between RM 1.8 and RM 2.2, four cores were analyzed. Peak concentrations were detected at depth in three of the four cores: LDW-SC34 (280 µg/kg dw in the 1-to-2-ft interval), LDW-SC38a (3,400 µg/kg dw in the 2-to-3-ft interval), and LDW-SC39 (440 µg/kg dw in the 1-to-2-ft interval). In the remaining core (LDW-SC35), the peak concentration was 370 µg/kg dw in the 0-to-2-ft interval.

In summary, peak concentrations in this section of the LDW were detected in the uppermost intervals and at depth. Peak concentrations were at depth for the locations with the highest subsurface concentrations (i.e., greater than the 95th percentile). One notable exception was in the navigation channel near RM 1.0, where the uppermost interval of the core had a peak concentration of 3,200 µg/kg dw. At other locations where the peak concentrations were in the uppermost interval of the core, concentrations were always less than or equal to 370 µg/kg dw.

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[69] Samples were collected from coarser depth intervals in the replicate core collected at this location (LDW-SC201); the peak concentration in this core was 1,450 µg/kg dw in the 0-to-1.5-ft interval.

RM 2.2 to RM 3.0

The highest interpolated total PCB concentrations (i.e., greater than the 95th percentile) in surface sediment between RM 2.2 and RM 3.0 were detected between RM 2.2 and RM 2.3 on the west side of the LDW in EAA 2 (up to 23,000 µg/kg dw), in the Slip 4 EAA (up to 25,000 µg/kg dw), and on the east side of the LDW between RM 2.9 and RM 3.0 in the Boeing Plant 2/Jorgensen Forge EAA (up to 16,000 µg/kg dw) (Map 4-26c). The highest concentrations in subsurface sediment were in Slip 4 (up to 35,000 µg/kg dw).

In the vicinity of EAA 2 between RM 2.2 and RM 2.3 on the western shoreline, two cores were collected. Core LDW-SC40 had low concentrations of total PCBs in the uppermost interval (160 µg/kg dw in the 0-to-1.3-ft interval) and PCBs were not detected below 1.3 ft. Core DR137, located just downstream from this area, had a peak PCB concentration of 730 µg/kg dw in the 2-to-4-ft interval, higher than the concentration of 181 µg/kg dw in the co-located surface sediment sample.

Interpolated total PCB concentrations in surface sediment along the western shoreline between RM 2.3 and RM 3.0 were relatively low (generally ≤ 110 µg/kg dw). Historical cores collected along the western shoreline between RM 2.4 and RM 2.8 had large composite intervals (generally 0 to 3 ft or 0 to 4 ft). The RI core with greater resolution in this area (LDW-SC46), near RM 2.7, had a peak total PCB concentration of 270 µg/kg dw in the 2-to-4-ft interval.

In the navigation channel between RM 2.2 and RM 2.7, total PCB concentrations in surface sediment were generally between the 25th and 75th percentiles. Historical cores in the navigation channel had larger intervals of 2 or 4 ft and had total PCB concentrations similar to interpolated concentrations in surface sediment in that area (with a maximum concentration of 270 µg/kg dw).

Along the eastern shoreline between RM 2.3 and RM 2.8, five cores were collected as part of the RI. Total PCB concentrations in two of the cores (LDW-SC42 and LDW-SC43) were relatively low (less than the 75th percentile). In the other cores (LDW-SC41, LDW-SC44, and LDW-SC45), peak total PCB concentrations were in lower core intervals.

Cores located at the mouth of Slip 4 had total PCB concentrations ranging from 320 to 980 µg/kg dw in the subsurface samples composited over 3 to 4 ft. However, the depth of the peak concentration was not discernable because of the large interval sampled. Seven cores at the head of Slip 4 and halfway through the slip generally had total PCB concentrations greater than the 95th percentile in at least one of the subsurface intervals. The peak total PCB concentration was in the 0-to-2-ft interval in three of the cores and in the 2-to-4-ft or 4-to-6-ft interval in the remaining four cores. High concentrations were detected in the subsurface intervals in these seven cores. For example, core SC01 had a peak concentration of 35,000 µg/kg dw in the 0-to-2-ft interval, and core SC03 had a peak concentration of 15,000 µg/kg dw in the 4-to-6-ft interval.

In cores collected from RM 2.9 to RM 3.0 on the eastern shoreline, data were not available from the uppermost 2 ft, so the location of the peak concentrations within those cores is not known. Concentrations in intervals between 2 and 5 ft were highly variable ranging from non-detect in cores DUW106, DUW138, and DUW109 to 9,200 µg/kg dw in core DUW 37.

In summary, total PCB concentrations were variable in this section of the LDW. Outside of Slip 4, in areas with sufficient data, peak concentrations were either low (≤ 160 µg/kg dw) or were detected at depth within the cores. Within Slip 4, concentrations were greater than the 95th percentile both at depth and in uppermost core intervals. Peak concentrations in subsurface cores from Slip 4 ranged from 990 to 35,000 µg/kg dw. The depths of peak concentrations in Slip 4 were highly variable.

RM 3.0 to RM 4.0

The highest total PCB concentrations (less than the 95th percentile) in surface sediment between RM 3.0 and RM 4.0 were detected between RM 3.5 and RM 3.9 on the east side of the LDW (up to 38,000 µg/kg dw in the Boeing Plant 2/Jorgensen Forge EAA), and on the west side of the LDW between RM 3.5 and RM 3.7 (up to 110,000 µg/kg dw) in the T-117 EAA (Map 4-26d).

Surface sediment concentrations were generally low (between the 25th and 50th percentiles) along the western shoreline from RM 3.0 to RM 3.5 (Map 4-26d). Subsurface samples in this area had relatively low total PCB concentrations, except for cores LDW-SC47 at RM 3.05 and SB-5 at RM 3.35; peak concentrations were detected in the 1-to-2-ft interval (2,000 µg/kg dw) and 2.5-to-5-ft interval (1,720 µg/kg dw), respectively, in these cores. Thus, total PCB concentrations were either low or the peak concentrations were detected at depth in this area.

In cores collected in the Boeing Plant 2/Jorgensen Forge EAA between RM 3.0 and RM 3.7 along the eastern shoreline, total PCB data from the uppermost 1- or 2-ft intervals were generally only available for cores collected between RM 3.5 and RM 3.7 (Maps 4-26e and 4-26f). Interpolated concentrations in surface sediment for most of this area were greater than the 95th percentile. In general, most core locations north of RM 3.6 with available subsurface data had peak concentrations in the uppermost intervals; these peak concentrations were greater than the 95th percentile concentration. South of RM 3.6, a number of cores had peak concentrations in deeper intervals; these peak concentrations were generally less than the 95th percentile. Fifty-six percent of the locations in the Boeing Plant 2/Jorgensen Forge EAA with co-located surface and subsurface sediment samples had total PCB concentrations in the surface sediment samples that were higher than the peak concentrations in the associated subsurface cores (see Maps 4-26e and 4-26f for concentrations in every sample).

In the 15 cores collected from the T-117 EAA between RM 3.5 and RM 3.7 on the western shoreline, the seven highest total PCB concentrations in subsurface sediment samples (ranging from 2,800 to 51,000 µg/kg dw) were all detected within the uppermost subsurface intervals (Map 4-26g) and 13 of the 15 cores in the T-117 EAA had the peak total PCB concentration in the uppermost interval. In most of these cores with co-located surface sediment data, the highest total PCB concentration was in the surface sediment (i.e., 9 out of 13 cores).

Peak total PCB concentrations in subsurface cores LDW-SC49a, SD-321-C, and SD-322-C, in the navigation channel between RM 3.5 and RM 3.7, were detected at depth (> 1 ft), with concentrations ranging from 750 to 2,770 µg/kg dw (Map 4-26d). All other cores in the navigation channel between RM 3.0 and RM 4.0 had low total PCB concentrations in subsurface intervals (≤ 110 µg/kg dw).

Surface sediment along the eastern shoreline between RM 3.7 and RM 3.9 generally had relatively high interpolated total PCB concentrations (between the 75th and 95th percentiles; Map 4-26d). In three cores in this area (LDW-SC51, DR220, and LDW-SC52), the peak concentrations were greater than the 95th percentile and were in the uppermost interval. In two of these cores, which had co-located surface sediment data, the concentrations were lowest in the surface sediment. In another core from this area at RM 3.8 (LDW-SC50a), the peak total PCB concentration (780 µg/kg dw) was in the 1-to-2-ft interval, although the concentration was higher (820 µg/kg dw) in the co-located surface sediment sample.

In summary, two EAAs with high total PCB concentrations are located within this reach. Vertical gradients were generally difficult to discern in the Boeing Plant 2/ Jorgensen Forge EAA, but in general the highest concentrations appeared to be in the uppermost intervals, particularly north of RM 3.6. In the T-117 EAA, peak concentrations were generally found in the uppermost intervals of the cores. Along the eastern shoreline south of RM 3.7, peak concentrations were generally in the uppermost intervals. In cores between RM 3.0 and RM 3.5, in the navigation channel and along the western shoreline, total PCB concentrations were generally low or peaks were detected at depth.

RM 4.0 to RM 5.0

Total PCB concentrations in surface and subsurface sediment between RM 4.0 and RM 5.0 were generally relatively low (less than the 50th percentile) (Map 4-26h), except for the eastern shoreline at RM 4.1 and the area between RM 4.8 and RM 5.0 (including the Norfolk EAA), where concentrations were greater than the 95th percentile in surface and subsurface sediment.

At RM 4.1 on the eastern shoreline, one small area had interpolated concentration in surface sediment greater than the 95th percentile; the only subsurface concentration greater than the 95th percentile near RM 4.1 on the eastern shoreline was also near this small area (2,500 µg/kg dw in core SH-04 in the only interval analyzed at 0.33 to

0.82 ft). In the area between RM 4.0 and RM 4.3 on the east side, including Slip 6, total PCB concentrations in subsurface sediments were generally less than the 75th percentile concentration. However,, only one relatively shallow subsurface core was analyzed at each of these locations, so it is not possible to determine the concentration gradient in deeper intervals at these locations.

Between RM 4.8 and RM 5.0, total PCB concentrations were variable. The total PCB concentration was low in core SC55 (13.5 µg/kg dw in the 0-to-1-ft interval), but concentrations were high in a co-located surface sediment sample (2,700 µg/kg dw). Within the Norfolk EAA, peak total PCB concentrations were 81,700 µg/kg dw in the 1-to-2-ft interval at NFK008 and 247 µg/kg dw in the 0-to-1-ft interval at nearby location NFK009. The area within the Norfolk EAA was dredged and backfilled with clean material after these cores were collected. The interpolated surface sediment data shown on Map 4-26h reflect post-dredging conditions.

Subsurface sediment cores were collected at two other locations between RM 4.0 and 5.0 as part of the RI. At one location (LDW-SC54), concentrations were low (less than the 50th percentile), and at the other location (LDW-SC56), the peak concentration (330 µg/kg dw) was detected in the uppermost interval of 0-to-2 ft. In summary, total PCB concentrations in both surface and subsurface sediments were generally lower in this section of the LDW than in downstream areas, except at one location near RM 4.1 and locations near the Norfolk EAA. Concentrations in all cores from the navigation channel were low.

4.2.3.4 Tissue

The RI tissue dataset contained data from 421 composite samples of fish and shellfish collected in the LDW between 1992 and 2007.[70] Over 74% of these samples were collected in 2004, 2005, and 2007 as part of the RI, as described in Section 4.1.1.3. Tissue data collected between 1992 and 2005 were used in the risk assessments; additional tissue data were collected in 2006 and 2007 after the risk assessments had been completed. The additional data from 2006 were collected by King County from Area T1 (Anchor and King County 2007). The additional data from 2007 were collected as part of the RI in all four tissue sampling areas (Map 4-10). This section presents total PCB data, as Aroclor sums and as a sum of individual PCB congener sums, and presents various analyses of these data.

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[70] An additional 45 whole-body concentrations calculated from concentrations in 10 English sole fillet and remainder samples and 35 crab edible meat and hepatopancreas samples were also included in the dataset (see Section 4.1.2).

Aroclor Data

Table 4-29 summarizes the total PCB (Aroclor sum) data in the RI tissue dataset. PCBs were detected in most of the tissue samples collected between 1992 and 2007, with a few exceptions.[71] In whole-body composite samples of fish, the lowest mean total PCB concentration was 71 µg/kg ww in juvenile chinook salmon collected between RM 0.5 and RM 0.9. The highest mean concentration in fish within a sampling area was 4,300 µg/kg ww in shiner surfperch in Area T2 (Table 4-29; Map 4-28). In general, mean concentrations in whole-body fish within individual sampling areas were highest in English sole and shiner surfperch (ranging from 300 to 3,900 µg/kg ww in English sole and from 270 to 4,300 µg/kg in shiner surfperch) and lowest in juvenile chinook salmon (ranging from 71 to 260 µg/kg ww).

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[71] Of all the tissue samples analyzed, PCBs were not detected in 4 of the 22 wild mussel samples, 19 of the 32 transplanted mussel samples, 1 of the 20 benthic invertebrate samples, and 3 of the 13 Dungeness crab edible-meat samples.

Table 4-29. Summary of composite tissue data for total PCBs (Aroclor sum)

		1992 – 2003				2004				2005				2006				2007			
			TOTAL PCBS (µg/kg ww)				TOTAL PCBS (µg/kg ww)				TOTAL PCBS (µg/kg ww)				TOTAL PCB (µg/kg ww)				TOTAL PCBS (µg/kg ww)		
TISSUE TYPE	AREA[a]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]
Fish, Whole Body																					
Juvenile chinook salmon	RM 0.5 – RM 0.9[c]	15	9.3 J	320 J	71	0	no data			0	no data			0	no data			0	no data		
	RM 2.6 – RM 2.9[d]	9	6.9 J	1,200	260	0	no data			0	no data			0	no data			0	no data		
English sole	T1	0	no data			6	2,700	4,700	3,700	6[e]	1,120	2,200	1,600	6	630	1,470	920	6	360	780	530
	T2	0	no data			6	3,300	4,200	3,900	6[e]	1,600	2,400	2,000	0	no data			6	380	980	690
	T3	0	no data			6	1,320	4,300	2,600	6[e]	610	2,200	1,400	0	no data			6	600	1,600 J	890
	T4	0	no data			3	1,640	1,800	1,700	3[e]	910	1,180	1,000	0	no data			1	300	300	300
Pacific staghorn sculpin	T1	0	no data			6	580	860	730	1	720 J	720 J	720	0	no data			0	no data		
	T2	0	no data			6	620	1,260	770	1	620	620	620	0	no data			0	no data		
	T3	0	no data			6	810	2,800	1,500	1	590	590	590	0	no data			0	no data		
	T4	0	no data			6	510	1,300	780	1	430	430	430	0	no data			0	no data		
Shiner surfperch	T1	3	350	620	500	6	970	1,830	1,400	6	530 J	960 J	780	7	250	730	457	6	200 J	360 J	270
	T2	0	no data			6	1,260	18,400 J	4,300	6	660	2,000	1,300	0	no data			6	290 J	610 J	420
	T3	0	no data			6	1,280	8,800	3,800	6	700	2,400	1,500	0	no data			6	430 J	1,330 J	760
	T4	0	no data			6	640	960	800	4	540	600	580	0	no data			4	260 J	410 J	320
Starry flounder	T4	0	no data			3	450	660	570	0	no data			0	no data			3	156	240	190
Fish, Fillet																					
English sole (with skin)	T1	0	no data			2	1,330	1,600	1,500	3	920	1,450	1,200	0	no data			3	260	500	340
	T2	0	no data			2	1,840	2,010	1,930	3	850	1,400	1,000	0	no data			3	170	360	290
	T3	0	no data			2	850	1,640	1,200	3	450	860	680	0	no data			3	340	490	400
	T4	0	no data			1	710	710	710	1	530	530	530	0	no data			0	no data		

Table 4-29, cont. Summary of composite tissue data for total PCBs (Aroclor sum)

TISSUE TYPE	AREA[a]	1992 – 2003				2004				2005				2006				2007			
			TOTAL PCBS (µg/kg ww)				TOTAL PCBS (µg/kg ww)				TOTAL PCBS (µg/kg ww)				TOTAL PCB (µg/kg ww)				TOTAL PCBS (µg/kg ww)		
		n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]
English sole (without skin)	T1 and RM 1.1 – RM 1.4[f]	12	79	360	190	0	no data			0	no data			0	no data			0	no data		
	RM 2.1 – RM 2.9	1	300 J	300 J	300	0	no data			0	no data			0	no data			0	no data		
	RM 2.9 – RM 3.6	1	280 J	280 J	280	0	no data			0	no data			0	no data			0	no data		
	RM 3.6 – RM 4.4	1	530 J	530 J	530	0	no data			0	no data			0	no data			0	no data		
Starry flounder (with skin)	T4	0	no data			1	450	450	450	0	no data			0	no data			1	63	63	63
Pile perch (with skin)	T3	0	no data			1	300	300	300	0	no data			0	no data			0	no data		
Striped perch (with skin)	RM 0.0 – RM 0.2	2	164 J	171 J	168	0	no data			0	no data			0	no data			0	no data		
	RM 4.0 – RM 4.1	0	no data			1	630	630	630	0	no data			0	no data			0	no data		
Crab, Edible Meat																					
Dungeness crab	T1	3	80 J	180	132	3	206 J	290	230	1	20 U	20 U	20 U	0				1	15	15	15
	T2	0	no data			0	no data			0	no data			0	no data			0	no data		
	T3	0	no data			3	212	300	250	1	20 U	20 U	20 U	0	no data			3	39	51 J	43
	T4	0	no data			1	240 J	240 J	240	1	20 U	20 U	20 U	0	no data			0	no data		
Red rock crab	T1	2	85	164	125	0	no data			0	no data			0	no data			0	no data		
Red rock/ Dungeness crab	T1	1	60	60	60	0	no data			0	no data			0	no data			0	no data		
Slender crab	T1	0	no data			3	210 J	390 J	270	0	no data			0	no data			3	41	48 J	43
	T2	0	no data			6	108	260 J	190	1	64	64	64	0	no data			3	27	46 J	38
	T3	0	no data			3	146	220	180	0	no data			0	no data			0	no data		
	T4	0	no data			0	no data			0	no data			0	no data			0	no data		

Table 4-29, cont. Summary of composite tissue data for total PCBs (Aroclor sum)

Tissue Type	Area[a]	1992 – 2003				2004				2005				2006				2007			
			Total PCBs (µg/kg ww)				Total PCBs (µg/kg ww)				Total PCBs (µg/kg ww)				Total PCB (µg/kg ww)				Total PCBs (µg/kg ww)		
		n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]
Crab, Hepatopancreas																					
Dungeness crab	T1	1	1,700	1,700	1,700	1	4,000	4,000	4,000	1	1,420	1,420	1,420	0	no data			1	280	280	280
	T2	0	no data			0	no data			0	no data			0	no data			0	no data		
	T3	0	no data			1	4,500	4,500	4,500	1	1,310	1,310	1,310	0	no data			3	420	1,020	650
	T4	0	no data			1	5,500	5,500	5,500	1	1,320	1,320	1,320	0	no data			0	no data		
Slender crab	T1	0	no data			1	1,490 J	1,490 J	1,490	0	no data			0	no data			3	480	660	540
	T2	0	no data			2	1,950 J	2,190 J	2,070	1	660	660	660	0	no data			3	250	270	260
	T3	0	no data			1	1,640	1,640	1,640	0	no data			0	no data			0	no data		
	T4	0	no data			0	no data			0	no data			0	no data			0	no data		
Crab, Whole Body (calc'd)[g]																					
Dungeness crab	T1	2	620 M	650 M	635	3	1,400 JM	1,400 JM	1,400	1	454 M	454 M	454	0	no data			1	97 M	97 M	97
	T2	0	no data			0	no data			0	no data			0	no data			0	no data		
	T3	0	no data			3	1,500 JM	1,600 JM	1,600	1	420 M	420 M	420	0	no data			3	160 M	351 JM	230
	T4	0	no data			1	1,900 JM	1,900 JM	1,900	1	423 M	423 M	423	0	no data			0	no data		
Slender crab	T1	0	no data			3	607 JM	731 JM	1650	0	no data			0	no data			3	180 M	240 JM	200
	T2	0	no data			6	729 JM	838 JM	776	1	250 M	250 M	250	0	no data			3	100 M	120 JM	110
	T3	0	no data			3	609 M	660 M	631	0	no data			0	no data			0	no data		
	T4	0	no data			0	no data			0	no data			0	no data			0	no data		
Invertebrates, Whole Body																					
Amphipods	RM 0.5 – RM 0.7	4	106	410	230	0	no data			0	no data			0	no data			0	no data		
Benthic invertebrates	LDW-wide[h]	0	no data			20	60 J	1,400	270	0	no data			0	no data			0	no data		
Shellfish																					
Clams, non-depurated	LDW-wide[h]	0	no data			14	24	580 J	140	0	no data			0	no data			6	15 J	310	110
Clams, depurated	LDW-wide[h]	0	no data			0	no data			0	no data			0	no data			6	14 J	270	98

Table 4-29, cont. Summary of composite tissue data for total PCBs (Aroclor sum)

		1992 – 2003				2004				2005				2006				2007			
			Total PCBs (µg/kg ww)				Total PCBs (µg/kg ww)				Total PCBs (µg/kg ww)				Total PCB (µg/kg ww)				Total PCBs (µg/kg ww)		
Tissue Type	Area[a]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]
Mussels, wild	RM 0.3 – RM 2.8[i]	22	16	60	34	0	no data			0	no data			0	no data			0	no data		
Mussels, transplanted	RM 0.5 – RM 1.2	32	35.9	73.1	27	0	no data			0	no data			0	no data			0	no data		

[a] Tissue sampling areas are shown on Maps 4-9 and 4-10.

[b] Mean concentration is the average of detected concentrations and one-half the RL for non-detected results. For non-detected results, the RL represents the highest RL for an individual Aroclor in that sample.

[c] Includes nine samples collected between RM 0.5 and RM 0.9 during the RI sampling event in 2003 and six samples collected between RM 0.8 and RM 0.9 during the East Waterway sampling event (both locations are within Area T1).

[d] These nine samples were collected between RM 2.6 and RM 2.9 (slightly downstream from Area T3) during the RI sampling event.

[e] A subset of these samples are calculated “whole-body” samples (three each from Areas T1, T2, and T3 and one from T4). Concentrations in “whole-body” samples were calculated using results from separate analyses of fillet and remainder composite samples (i.e., all remaining tissue and fluids after fillets were removed from the specimens). The calculated English sole “whole-body” concentrations were based on the relative weights of and total PCB concentrations in skin-on fillet and remainder tissues collected in 2005.

[f] Includes data from three samples collected between RM 1.1 and RM 1.4 in 1995 (slightly upstream from Area T1) (Battelle 1996; Frontier Geosciences 1996), and nine samples collected between RM 0.5 and RM 1.0 (within Area T1) during the PSAMP and King County WQA studies between 1992 and 1997 (West et al. 2001; King County 1999e).

[g] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Total PCB concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data).

[h] The benthic invertebrate and clam samples were collected throughout the LDW (Map 4-10).

[i] All but three of the wild mussel samples were collected between RM 0.3 and 1.2; the remaining three samples were collected at one sampling location at the mouth of Slip 4 near RM 2.8.

J – estimated concentration
JM – calculated from an estimated concentration
M – calculated concentration
n – number of samples
PCB – polychlorinated biphenyl
PSAMP – Puget Sound Ambient Monitoring Program
RM – river mile
U – not detected at RL shown
WQA – water quality assessment
ww – wet weight

One shiner surfperch composite sample collected from Subarea T2E in 2004 had a total PCB concentration based on the sum of Aroclors (18,400 µg/kg ww) that was much higher than any other tissue samples in the LDW.[72] The total PCB concentration in this sample based on the sum of PCB congeners was also very high (12,230 µg/kg ww), providing confirmation of the Aroclor result. To better understand the variability of total PCB concentrations in shiner surfperch collected from Subarea T2E in 2004, 10 archived fish from this subarea were analyzed individually.[73] Total PCB concentrations in these individual fish ranged from 172 to 1,140 µg/kg ww, with a mean concentration of 640 µg/kg ww. Based on these data, it is likely that one or more of the 10 fish included in the composite sample with 18,400 µg/kg ww had a very high concentration of total PCBs.

Fillet samples from four different fish species were also analyzed. The largest fillet dataset for PCBs is for English sole, with 15 composite samples (without skin) collected between 1992 and 1998 and 26 composite samples (with skin) collected between 2004 and 2007. Other species with available fillet data are pile perch (one composite sample), starry flounder (one composite sample), and striped perch (three composite samples); all of these fillet samples were analyzed with skin. In fillet samples collected in the 1990s without skin, total PCB tissue concentrations generally were lower than those in samples collected between 2004 and 2007 with skin (79 to 530 µg/kg ww and 290 to 2,010 µg/kg ww, respectively). This difference may be related to the year of sampling (1992 to 1998 vs. 2004 to 2007), the absence of skin,[74] or other factors.

Mean total PCB concentrations within all sampling areas for crab edible meat, benthic invertebrates, and clams were 270 µg/kg ww or lower; in general, these concentrations were lower than those in fish (Table 4-29). In crabs, mean total PCB concentrations in edible-meat composite samples from the four tissue sampling areas were much lower (non-detected to 270 µg/kg ww) than those in hepatopancreas composite samples (260 to 5,500 µg/kg ww) (Table 4-29). Mean concentrations in invertebrate samples (amphipods collected from RM 0.5 to RM 0.7 and benthic invertebrates collected from

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[72] Using Rosner's test for outliers from a log-normal distribution, this value was considered to be a statistical outlier ($p < 0.005$).

[73] A total of 20 shiner surfperch (> 80 mm) were collected in subarea T2E in 2004. Ten of these fish were included in the initial composite sample for this area, and the 10 remaining fish were archived frozen as individual fish.

[74] The potential influence of skin on the total PCB concentrations in the 2004 English sole fillet samples was investigated to facilitate comparison of 2004 samples with historical data. One fillet with skin and one fillet without skin were created from each of 10 individual fish archived from Area T1, for a total of 20 samples. Total PCB concentrations in the fillets with skin (420 to 1,440 µg/kg ww) and in the fillets without skin (230 to 1,570 µg/kg ww) were not significantly different when tested with analysis of variance ($p = 0.47$).

throughout the LDW) were 230 and 270 µg/kg ww, respectively. These invertebrate tissues, as well as soft-shell clam tissues collected from throughout the LDW (140 µg/kg\ ww), were generally lower than fish tissues. Some of the lowest mean total PCB concentrations in any of the tissue types were detected in wild and transplanted mussels collected primarily from RM 0.3 to RM 1.2 (34 and 27 µg/kg, respectively). Mean total PCB concentrations in Dungeness crab edible meat samples collected in 2005 (20 U µg/kg ww) and 2007 (15 to 43 µg/kg ww) were lower than in mussels, but mean total PCB concentrations in Dungeness crab edible meat samples in 2004 (230 to 250 µg/kg ww) were higher than in mussels.

Mean lipid contents in whole-body fish and crab tissue were higher in English sole and shiner surfperch (3.67 to 7.1% and 2.8 to 6.38%, respectively) than in Pacific staghorn sculpin and crabs (1.18 to 2.3% and 0.928 to 5.4% respectively) (Table 4-30). Mean lipid contents were lower in English sole fillet tissues (1.7 to 4.0%) than in English sole whole-body tissues (3.68 to 7.1%), and mean lipid contents in crab edible-meat tissues (0.15 to 2.0%) were lower than in hepatopancreas tissues (1.9 to 13%). Lipid contents were relatively low in benthic invertebrates (0.89 to 3.0%), clams (0.71 to 0.87%), and mussels (0.89 to 1.3%).

Mean lipid-normalized total PCB concentrations in fish tissue ranged from 2.8 to 100 mg/kg-lipid, with no clear differences in concentrations among fish species or tissue types (i.e., whole body vs. fillet) (Table 4-31). Mean lipid-normalized total PCB concentrations in crab tissue ranged from non-detected to 87 mg/kg-lipid; there were no clear differences among crab species or tissue types (i.e., hepatopancreas or edible meat) (Table 4-31). Mean lipid-normalized total PCB concentrations ranged from 12 to 41 mg/kg-lipid in benthic invertebrates, 11 to 21 mg/kg-lipid in clams, and 3.3 to 4.5 mg/kg-lipid in mussels.

Table 4-30. Summary of composite tissue data for lipids

TISSUE TYPE	AREA[a]	1992 – 2003				2004				2005				2006				2007			
			LIPIDS (% ww)				LIPIDS (% ww)				LIPIDS (% ww)				LIPIDS (% ww)				LIPIDS (% ww)		
		n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]
Fish, Whole Body																					
Juvenile chinook salmon	RM 0.5 – RM 0.9[b]	15	0.55	3.5	2.0	0	no data			0	no data			0	no data			0	no data		
	RM 2.6 – RM 2.9[c]	9	0.74	2.8	1.6	0	no data			0	no data			0	no data			0	no data		
English sole	T1	0	no data			6	4.3	6.8	5.6	6[d]	3.13	6.14	4.74	6	2.46	5.06	3.67	6	3.83	7.22	6.04
	T2	0	no data			6	5.7	8.7	7.1	6[d]	4.89	6.83	5.74	0	no data			6	4.46	9.00	6.44
	T3	0	no data			6	2.6	6.6	4.9	6[d]	3.09	7.53	5.19	0	no data			6	2.34	10.9	6.44
	T4	0	no data			3	4.8	6.2	5.6	3[d]	3.85	6.25	5.24	0	no data			1	4.62	4.62	4.62
Pacific staghorn sculpin	T1	0	no data			6	1.8	2.4	2.2	1	2.17	2.17	2.17	0	no data			0	no data		
	T2	0	no data			6	1.8	2.7	2.3	1	1.92	1.92	1.92	0	no data			0	no data		
	T3	0	no data			6	1.8	2.1	1.9	1	1.34	1.34	1.34	0	no data			0	no data		
	T4	0	no data			6	1.3	2.5	2.1	1	1.18	1.18	1.18	0	no data			0	no data		
Shiner surfperch	T1	3	1.6	4.0	2.8	6	2.3	5.0	3.4	6	4.31	6.16	5.58	7	2.79	7.32	5.17	6	1.80	4.94	3.13
	T2	0	no data			6	2.5	5.6	4.0	6	4.74	5.99	5.43	0	no data			6	2.90	4.86	4.04
	T3	0	no data			6	3.1	5.6	4.0	6	4.98	6.92	5.78	0	no data			6	3.43	4.94	4.14
	T4	0	no data			6	3.0	5.6	4.1	4	6.16	6.93	6.38	0	no data			4	3.62	4.78	4.33
Starry flounder	T4	0	no data			3	2.1	2.5	2.2	0	no data			0	no data			3	0.917	3.29	1.95
Fish, Fillet																					
English sole (with skin)	T1	0	no data			2	2.6	3.1	2.9	3	2.95	5.09	3.69	0	no data			3	2.85	4.11	3.32
	T2	0	no data			2	3.4	4.3	3.9	3	3.42	4.62	3.98	0	no data			3	2.14	3.63	2.97
	T3	0	no data			2	1.6	3.6	2.6	3	2.04	4.12	3.14	0	no data			3	1.77	3.26	2.66
	T4	0	no data			1	1.7	1.7	1.7	1	2.82	2.82	2.82	0	no data			0	no data		

Table 4-30, cont. Summary of composite tissue data for lipids

		1992 – 2003				2004				2005				2006				2007			
			Lipids (% ww)				Lipids (% ww)				Lipids (% ww)				Lipids (% ww)				Lipids (% ww)		
Tissue Type	Area[a]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]
English sole (without skin)	T1 and RM 1.1 – RM 1.4[g]	12	0.24	12	3.0	0	no data			0	no data			0	no data			0	no data		
	RM 2.1 – RM 2.9	0	no data			0	no data			0	no data			0	no data			0	no data		
	RM 2.9 – RM 3.6	0	no data			0	no data			0	no data			0	no data			0	no data		
	RM 3.6 – RM 4.4	0	no data			0	no data			0	no data			0	no data			0	no data		
Starry flounder (with skin)	T4	0	no data			1	2.6	2.6	2.6	0	no data			0	no data			1	2.23	2.23	2.23
Pile perch (with skin)	T3	0	no data			1	1.1	1.1	1.1	0	no data			0	no data			0	no data		
Striped perch (with skin)	RM 0.0 – RM 0.2	0	no data			0	no data			0	no data			0	no data			0	no data		
	RM 4.0 – RM 4.1	0	no data			1	1.4	1.4	1.4	0	no data			0	no data			0	no data		
Crab, Edible Meat																					
Dungeness crab	T1	2	1.6	2.4	2.0	3	0.28	0.39	0.34	1	0.191	0.191	0.191	0	no data			1	0.440	0.440	0.440
	T2	0	no data			0	no data			0	no data			0	no data			0	no data		
	T3	0	no data			3	0.23	0.47	0.37	1	0.146	0.146	0.146	0	no data			3	0.508	0.644	0.561
	T4	0	no data			1	0.72	0.72	0.72	1	0.232	0.232	0.232	0	no data			0	no data		
Red rock crab	T1	0	no data			0	no data			0	no data			0	no data			0	no data		
Red rock/ Dungeness crab	T1	0	no data			0	no data			0	no data			0	no data			0	no data		
Slender crab	T1	0	no data			3	0.43	0.74	0.57	0	no data			0	no data			3	0.408	0.444	0.427
	T2	0	no data			6	0.23	0.47	0.36	1	0.315	0.315	0.315	0	no data			3	0.452	0.628	0.557
	T3	0	no data			3	0.45	0.52	0.49	0	no data			0	no data			0	no data		
	T4	0	no data			0	no data			0	no data			0	no data			0	no data		

Table 4-30, cont. Summary of composite tissue data for lipids

Tissue Type	Area[a]	1992 – 2003				2004				2005				2006				2007			
			Lipids (% ww)				Lipids (% ww)				Lipids (% ww)				Lipids (% ww)				Lipids (% ww)		
		n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]
Crab, Hepatopancreas																					
Dungeness crab	T1	1	13	13	13	1	4.6	4.6	4.6	1	8.14	8.14	8.14	0	no data			1	3.72	3.72	3.72
	T2	0	no data			0	no data			0	no data			0	no data			0	no data		
	T3	0	no data			1	6.3	6.3	6.3	1	4.28	4.28	4.28	0	no data			3	4.56	6.87	5.81
	T4	0	no data			1	7.9	7.9	7.9	1	5.52	5.52	5.52	0	no data			0	no data		
Slender crab	T1	0	no data			1	1.9	1.9	1.9	0	no data			0	no data			3	1.64	2.79	2.05
	T2	0	no data			2	2.7	3.6	3.2	1	2.47	2.47	2.47	0	no data			3	3.07	4.10	3.69
	T3	0	no data			1	2.2	2.2	2.2	0	no data			0	no data			0	no data		
	T4	0	no data			0	no data			0	no data			0	no data			0	no data		
Crab, Whole Body (calc'd)[g]																					
Dungeness crab	T1	2	5.1	5.7	5.4	3	1.6	1.7	1.7	1	2.66	2.66	2.66	0	no data			1	1.46	1.46	1.46
	T2	0	no data			0	no data			0	no data			0	no data			0	no data		
	T3	0	no data			3	2.1	2.3	2.2	1	1.43	1.43	1.43	0	no data			3	1.76	2.50	2.19
	T4	0	no data			1	2.9	2.9	2.9	1	1.87	1.87	1.87	0	no data			0	no data		
Slender crab	T1	0	no data			3	0.89	1.1	0.98	0	no data			0	no data			3	0.790	1.17	0.928
	T2	0	no data			6	1.0	1.4	1.2	1	0.983	0.983	0.983	0	no data			3	1.39	1.62	1.53
	T3	0	no data			3	0.99	1.0	1.0	0	no data			0	no data			0	no data		
	T4	0	no data			0	no data			0	no data			0	no data			0	no data		
Invertebrates, Whole Body																					
Amphipods	RM 0.5 – RM 0.7	4	0.66	5.3	3.0	0	no data			0	no data			0	no data			0	no data		
Benthic invertebrates	LDW-wide[g]	0	no data			20	0.35	1.4	0.89	0	no data			0	no data			0	no data		
Shellfish																					
Clams, non-depurated	LDW-wide[g]	0	no data			14	0.55	0.83	0.71	0	no data			0	no data			15	0.599	1.03	0.788
Clams, depurated	LDW-wide[g]	0	no data			0	no data			0	no data			0	no data			15	0.619	1.10	0.866

Table 4-30, cont. Summary of composite tissue data for lipids

		1992 – 2003				2004				2005				2006				2007			
			Lipids (% ww)				Lipids (% ww)				Lipids (% ww)				Lipids (% ww)				Lipids (% ww)		
Tissue Type	Area[a]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]
Mussels, wild	RM 0.3 – RM 2.8[h]	21	0.29	2.4	0.89	0	no data			0	no data			0	no data			0	no data		
Mussels, transplanted	RM 0.5 – RM 1.2	32	0.25	2.6	1.3	0	no data			0	no data			0	no data			0	no data		

[a] Tissue sampling areas are shown on Maps 4-9 and 4-10.

[b] Includes nine samples collected between RM 0.5 and RM 0.9 during the RI sampling event in 2003 and six samples collected between RM 0.8 and RM 0.9 during the East Waterway sampling event in 2002 (both locations are within Area T1).

[c] These nine samples were collected between RM 2.6 and RM 2.9 (slightly downstream from Area T3) during the RI sampling event.

[d] A subset of these samples are calculated “whole-body” samples (three each from Areas T1, T2, and T3 and one from T4). Concentrations in “whole-body” samples were calculated using results from separate analyses of fillet and remainder composite samples (i.e., all remaining tissue and fluids after fillets were removed from the specimens). The calculated English sole “whole-body” concentrations were based on the relative weights of and total PCB concentrations in skin-on fillet and remainder tissues collected in 2005.

[e] Includes data from three samples collected between RM 1.1 and RM 1.4 (slightly upstream from Area T1) in 1995 (Battelle 1996; Frontier Geosciences 1996),, and nine samples collected between RM 0.5 and RM 1.0 (within Area T1) during the PSAMP and King County WQA studies between 1992 and 1997 (West et al. 2001; King County 1999e).

[f] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Total PCB concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data).

[g] The benthic invertebrate and clam samples were collected throughout the LDW (Map 4-10).

[h] All but three of the wild mussel samples were collected between RM 0.3 and 1.2; the remaining three samples were collected at one sampling location at the mouth of Slip 4 near RM 2.8.

LDW – Lower Duwamish Waterway
n – number of samples
PCB – polychlorinated biphenyl
PSAMP – Puget Sound Ambient Monitoring Program
RI – remedial investigation
RL – reporting limit
RM – river mile
WQA – water quality assessment
ww – wet weight

Table 4-31. Summary of composite tissue data for total PCBs (Aroclor sum) on a lipid-normalized basis

		1992 – 2003				2004				2005				2006				2007			
			TOTAL PCBs (mg/kg lipid)				TOTAL PCBs (mg/kg lipid)				TOTAL PCBs (mg/kg lipid)				TOTAL PCBs (mg/kg lipid)				TOTAL PCBs (mg/kg lipid)		
TISSUE TYPE	AREA[a]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]
Fish, Whole Body																					
Juvenile chinook salmon	RM 0.5 – RM 0.9[c]	15	0.46	35 J	7.1	0	no data			0	no data			0	no data			0	no data		
	RM 2.6 – RM 2.9[d]	9	0.25 J	160	29	0	no data			0	no data			0	no data			0	no data		
English sole	T1	0	no data			6	58	75	65	6[e]	27.8 M	52.1	36	6	17	39.9	26	6	5.0	11	8.9
	T2	0	no data			6	48	59	55	6[e]	28.0 M	45	34	0	no data			6	8.5	17	11
	T3	0	no data			6	40	78	52	6[e]	18 M	36.8	26	0	no data			6	8.4 J	28 J	16
	T4	0	no data			3	27	34	30	3[e]	15 M	24	20	0	no data			1	6.5	6.5	6.5
Pacific staghorn sculpin	T1	0	no data			6	27 J	42 J	33	1	33 J	33 J	33	0	no data			0	no data		
	T2	0	no data			6	24 J	55	34	1	32	32	32	0	no data			0	no data		
	T3	0	no data			6	40	160	81	1	44	44	44	0	no data			0	no data		
	T4	0	no data			6	28	53	37	1	36	36	36	0	no data			0	no data		
Shiner surfperch	T1	3	12	33	20	6	19	55	44	6	8.6 J	22	14	7	4.7	14	9.2	6	6.8 J	14 J	9.2
	T2	0	no data			6	33	330 J	91	6	13	42	24	0	no data			6	6.6 J	14 J	10
	T3	0	no data			6	35	284	100	6	13	35	25	0	no data			6	11 J	26.9 J	18
	T4	0	no data			6	16	29	20	4	8.4	9.7	9.1	0	no data			4	5.4 J	8.6 J	7.3
Starry flounder	T4	0	no data			3	21	31	26	0	no data			0	no data			3	7.3	19	12
Fish, Fillet																					
English sole (with skin)	T1	0	no data			2	51	52	51	3	28.5	39.0	33	0	no data			3	9.0	12	10
	T2	0	no data			2	43	59	51	3	23	30	26	0	no data			3	7.9	11	9.6
	T3	0	no data			2	46	53	49	3	21	22	22	0	no data			3	13	19	16
	T4	0	no data			1	42	42	42	1	19	19	19	0	no data			0	no data		

Table 4-31, cont. Summary of composite tissue data for total PCBs (Aroclor sum) on a lipid-normalized basis

		1992 – 2003				2004				2005				2006				2007			
			Total PCBs (mg/kg lipid)				Total PCBs (mg/kg lipid)				Total PCBs (mg/kg lipid)				Total PCBs (mg/kg lipid)				Total PCBs (mg/kg lipid)		
Tissue Type	Area[a]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]
English sole (without skin)	T1 and RM 1.1 – RM 1.4[f]	12	1.5	130	42	0	no data			0	no data			0	no data			0	no data		
	RM 2.1 – RM 2.9	1	no lipids data			0	no data			0	no data			0	no data			0	no data		
	RM 2.9 – RM 3.6	1	no lipids data			0	no data			0	no data			0	no data			0	no data		
	RM 3.6 – RM 4.4	1	no lipids data			0	no data			0	no data			0	no data			0	no data		
Starry flounder (with skin)	T4	0	no data			1	17	17	17	0	no data			0	no data			1	2.8	2.8	2.8
Pile perch (with skin)	T3	0	no data			1	27	27	27	0	no data			0	no data			0	no data		
Striped perch (with skin)	RM 0.0 – RM 0.2	2	no lipids data			0	no data			0	no data			0	no data			0	no data		
	RM 4.0 – RM 4.1	0	no data			1	45	45	45	0	no data			0	no data			0	no data		
Crab, Edible Meat																					
Dungeness crab	T1	2	5.6	11	8.3	3	53 J	100	71	1	10 U	10 U	10 U	0	no data			1	3.4	3.4	3.4
	T2	0	no data			0	no data			0	no data			0	no data			0	no data		
	T3	0	no data			3	45	98	73	1	14 U	14 U	14 U	0	no data			3	6.2	9.6 J	7.8
	T4	0	no data			1	33 J	33 J	33	1	8.6 U	8.6 U	8.6 U	0	no data			0	no data		
Red rock crab	T1	2	no lipids data			0	no data			0	no data			0	no data			0	no data		
Red rock/ Dungeness crab	T1	1	no lipids data			0	no data			0	no data			0	no data			0	no data		
Slender crab	T1	0	no data			3	41	53 J	48	0	no data			0	no data			3	9.2	12 J	10
	T2	0	no data			6	23	78	58	1	20	20	20	0	no data			3	6.0	7.3 J	6.7
	T3	0	no data			3	28	49	37	0	no data			0	no data			0	no data		
	T4	0	no data			0	no data			0	no data			0	no data			0	no data		

Table 4-31, cont. Summary of composite tissue data for total PCBs (Aroclor sum) on a lipid-normalized basis

		1992 – 2003				2004				2005				2006				2007			
			Total PCBs (mg/kg lipid)				Total PCBs (mg/kg lipid)				Total PCBs (mg/kg lipid)				Total PCBs (mg/kg lipid)				Total PCBs (mg/kg lipid)		
Tissue Type	Area[a]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]
Crab, Hepatopancreas																					
Dungeness crab	T1	1	13	13	13	1	87	87	87	1	17.4	17.4	17.4	0	no data			1	7.5	7.5	7.5
	T2	0	no data			0	no data			0	no data			0	no data			0	no data		
	T3	0	no data			1	71	71	71	1	30.6	30.6	30.6	0	no data			3	8.7	14.8	11
	T4	0	no data			1	70	70	70	1	23.9	23.9	23.9	0	no data			0	no data		
Slender crab	T1	0	no data			1	78 J	78 J	78	0	no data			0	no data			3	17	40	28
	T2	0	no data			2	61 J	72 J	67	1	27	27	27	0	no data			3	6.4	8.8	7.3
	T3	0	no data			1	75	75	75	0	no data			0	no data			0	no data		
	T4	0	no data			0	no data			0	no data			0	no data			0	no data		
Crab, Whole Body (calc'd)[g]																					
Dungeness crab	T1	2	11 M	13 M	12	3	82 JM	88 M	84	1	17.1 M	17.1 M	17.1	0	no data			1	6.6 M	6.6 M	6.6
	T2	0	no data			0	no data			0	no data			0	no data			0	no data		
	T3	0	no data			3	65 M	76 M	71	1	29 M	29 M	29	0	no data			3	8.3 M	14.0 JM	11
	T4	0	no data			1	66 JM	66 JM	66	1	22.6 M	22.6 M	22.6	0	no data			0	no data		
Slender crab	T1	0	no data			3	64 JM	68 JM	66	0	no data			0	no data			3	15 M	30 JM	22
	T2	0	no data			6	54 JM	73 JM	65	1	25 M	25 M	25	0	no data			3	6.3 M	8.6 JM	7.2
	T3	0	no data			3	61 M	67 M	63	0	no data			0	no data			0	no data		
	T4	0	no data			0	no data			0	no data			0	no data			0	no data		
Invertebrates, Whole Body																					
Amphipods	RM 0.5 – RM 0.7	4	5.3	17	12	0	no data			0	no data			0	no data			0	no data		
Benthic invertebrates	LDW-wide[h]	0	no data			20	6.5 J	400	41	0	no data			0	no data			0	no data		
Shellfish																					
Clams, non-depurated	LDW-wide[h]	0	no data			14	4.0	73 J	21	0	no data			0	no data			6	2.0 J	40	14
Clams, depurated	LDW-wide[h]	0	no data			0	no data			0	no data			0	no data			6	1.7 J	29	11

Table 4-31, cont. Summary of composite tissue data for total PCBs (Aroclor sum) on a lipid-normalized basis

TISSUE TYPE	AREA[a]	1992 – 2003				2004				2005				2006				2007			
			TOTAL PCBs (mg/kg lipid)				TOTAL PCBs (mg/kg lipid)				TOTAL PCBs (mg/kg lipid)				TOTAL PCBs (mg/kg lipid)				TOTAL PCBs (mg/kg lipid)		
		n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]
Mussels, wild	RM 0.3 – RM 2.8[i]	21	2.0	12	4.5	0	no data			0	no data			0	no data			0	no data		
Mussels, transplanted	RM 0.5 – RM 1.2	30	3.4	22	3.3	0	no data			0	no data			0	no data			0	no data		

[a] Tissue sampling areas are shown on Maps 4-9 and 4-10.

[b] Mean concentration is the average of detected lipid-normalized total PCB concentrations and one-half the lipid-normalized RL for non-detected results. For non-detected results, the RL represents the highest RL for an individual Aroclor in that sample.

[c] Includes nine samples collected between RM 0.5 and RM 0.9 during the RI sampling event in 2003 and six samples collected between RM 0.8 and RM 0.9 during the East Waterway sampling event in 2002 (both locations are within Area T1).

[d] These nine samples were collected between RM 2.6 and RM 2.9 (slightly downstream from Area T3) during the RI sampling event.

[e] Includes calculated “whole-body” samples (three each from Areas T1, T2, and T3 and one from T4). Concentrations in “whole-body” samples were calculated using results from separate analyses of fillet and remainder composite samples (i.e., all remaining tissue and fluids after fillets were removed from the specimens). The calculated English sole “whole-body” concentrations were based on the relative weights of and total PCB concentrations in skin-on fillet and remainder tissues collected in 2005.

[f] A subset of these samples are data from three samples collected between RM 1.1 and RM 1.4 (slightly upstream from Area T1) in 1995 (Battelle 1996; Frontier Geosciences 1996), and nine samples collected between RM 0.5 and RM 1.0 (within Area T1) during the PSAMP and King County WQA studies between 1992 and 1997 (West et al. 2001; King County 1999e).

[g] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Total PCB concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data).

[h] The benthic invertebrate and clam samples were collected throughout the LDW (Map 4-10).

[i] All but three of the wild mussel samples were collected between RM 0.3 and 1.2; the remaining three samples were collected at one sampling location at the mouth of Slip 4 near RM 2.8.

J – estimated concentration
LDW – Lower Duwamish Waterway
M – calculated concentration
n – number of samples
PCB – polychlorinated biphenyl
PSAMP – Puget Sound Ambient Monitoring Program
RI – remedial investigation
RL – reporting limit
RM – river mile
U – not detected at RL shown
WQA – water quality assessment

Tissue samples collected in 2004 were analyzed for PCB Aroclors by Columbia Analytical Services, Inc. (CAS); samples collected in 2005, 2006, and 2007 were analyzed by Analytical Resources, Inc. (ARI). The 2004 samples were reported as a mixture of Aroclors 1248, 1254, and 1260 (Table 4-32). In 2005, 2006, and 2007, the samples were reported as a mixture of Aroclors 1254 and 1260 (Aroclor 1248 was not detected), except for one shiner surfperch sample collected in 2005 (from Area T2-E) with a detected concentration of Aroclor 1248. Based on these data, the PCB mixtures in samples collected from 2005 to 2007 appear to have had a higher proportion of the more highly chlorinated PCB congeners than those collected in 2004.

Table 4-32. Aroclor composition of 2004 through 2007 fish tissue samples

			Mean Percent Composition of Total PCBs (SD)		
Sample Type	Year	n	Aroclor 1248	Aroclor 1254	Aroclor 1260
English sole – fillets	2004	7	19.2 (1.41)	40.3 (1.11)	40.5 (0.95)
	2005	10	nd	68.9 (2.19)	31.1 (2.19)
English sole – whole body	2004	21	19.2 (2.95)	41.2 (1.28)	39.6 (3.64)
	2005	21	nd	63.3 (5.52)	36.7 (5.52)
	2006	6	nd	49.1 (2.05)	50.9 (2.05)
	2007	19	nd	59.6 (4.95)	40.4 (4.95)
Shiner surfperch	2004	24	15.0 (6.19)	36.3 (7.93)	48.8 (13.62)
	2005	22	0.29 (1.21)	57.9 (9.65)	42.2 (8.65)
	2006	7	nd	44.2 (4.77)	55.8 (4.77)
	2007	22	nd	43.0 (6.33)	57 (6.33)
Pacific staghorn sculpin	2004	24	13.9 (6.31)	37.4 (2.93)	48.7 (8.86)
	2005	4	nd	54.6 (8.72)	45.4 (8.72)

n – number of composite samples
nd – not detected
PCB – polychlorinated biphenyl
SD – standard deviation

PCB Congener Data

PCB congener data are available for all tissue types except amphipods, mussels, red rock crab, and juvenile chinook salmon. All 209 PCB congeners were analyzed in a subset of all tissue types sampled in 2004, 2005, and 2007 as part of the RI. PCB congener data are also available for three English sole fillet samples from the 1995 Elliott Bay/Duwamish River fish tissue investigation (EVS unpublished). This section evaluates differences in the relative abundances of individual PCB congeners within a tissue type and among years. In addition, a summary of EPA's analysis of the relative abundance of the different PCB homolog groups in LDW sediments and tissues is presented.

The relative abundances of PCB congeners in each tissue type were calculated to determine if patterns were different in different species with different exposure regimes.

The first step in examining relative abundances of individual PCB congeners within a tissue type and among years (2004, 2005, and 2007) was to condense the list of 209 PCB congeners to a smaller list based on the PCB congeners that contributed most to the total PCB concentration in each sample. These PCB congeners were identified by calculating the percentage of the total PCB concentration represented by each PCB congener.[75]

A PCB congener was selected for inclusion in plots showing relative abundances of individual PCB congeners within a sample if the PCB congener was detected in every sample among all tissue types and if the concentration comprised at least 2% of the total PCB concentration in any single sample. Twenty-eight PCB congeners were identified using these criteria. Patterns in the relative abundances of PCB congeners among species were evaluated based on average relative abundance values for each PCB congener in each tissue type using available data collected in 2004, 2005, and 2007 for fish, benthic invertebrates, clams, and crabs. The sum of the concentrations of these 28 PCB congeners represented 76 to 84% of the total PCB concentrations in tissue samples.

The PCB congener patterns for shiner surfperch from each of the sampling areas were compared for 2004, 2005, and 2007 (Figure 4-5). The shiner surfperch composite samples collected in 2004 had the greatest variability in PCB congener patterns among individual composite samples. Two of the nine 2004 shiner surfperch composite samples (collected from subareas T3-E and T3-F) had higher contributions from the more highly chlorinated PCB congeners (PCB-180, PCB-183, and PCB-187) (Figure 4-5). One shiner surfperch composite sample (collected from subarea T2-E) had higher relative contributions from the less chlorinated PCB congeners (PCB-044, PCB-049, and PCB-052) (Figure 4-5). In general, PCB congener patterns for shiner surfperch composite samples in Areas T1 and T2 tended to contain a greater proportion of more highly chlorinated PCB congeners in 2007 and 2005 compared with 2004. There do not appear to be consistent differences in the PCB congener pattern over time in Area T3. Area T4 was sampled in 2004 only.

The PCB congener patterns in English sole showed higher contributions from the more highly chlorinated PCB congeners (PCB-153, PCB-180 and PCB-187) in Area T1 in 2005 and 2007 than in 2004 (Figure 4-6). The PCB congener patterns in Areas T2 and T3 were consistent for all three years. The PCB congener patterns for the composite Pacific staghorn sculpin samples (Figure 4-7) were consistent with the PCB congener patterns for English sole in 2004 (Figure 4-6). English sole samples collected from Area T4 were only analyzed for PCB congeners in 2004; Pacific staghorn sculpin were only collected in Area T4 in 2004.

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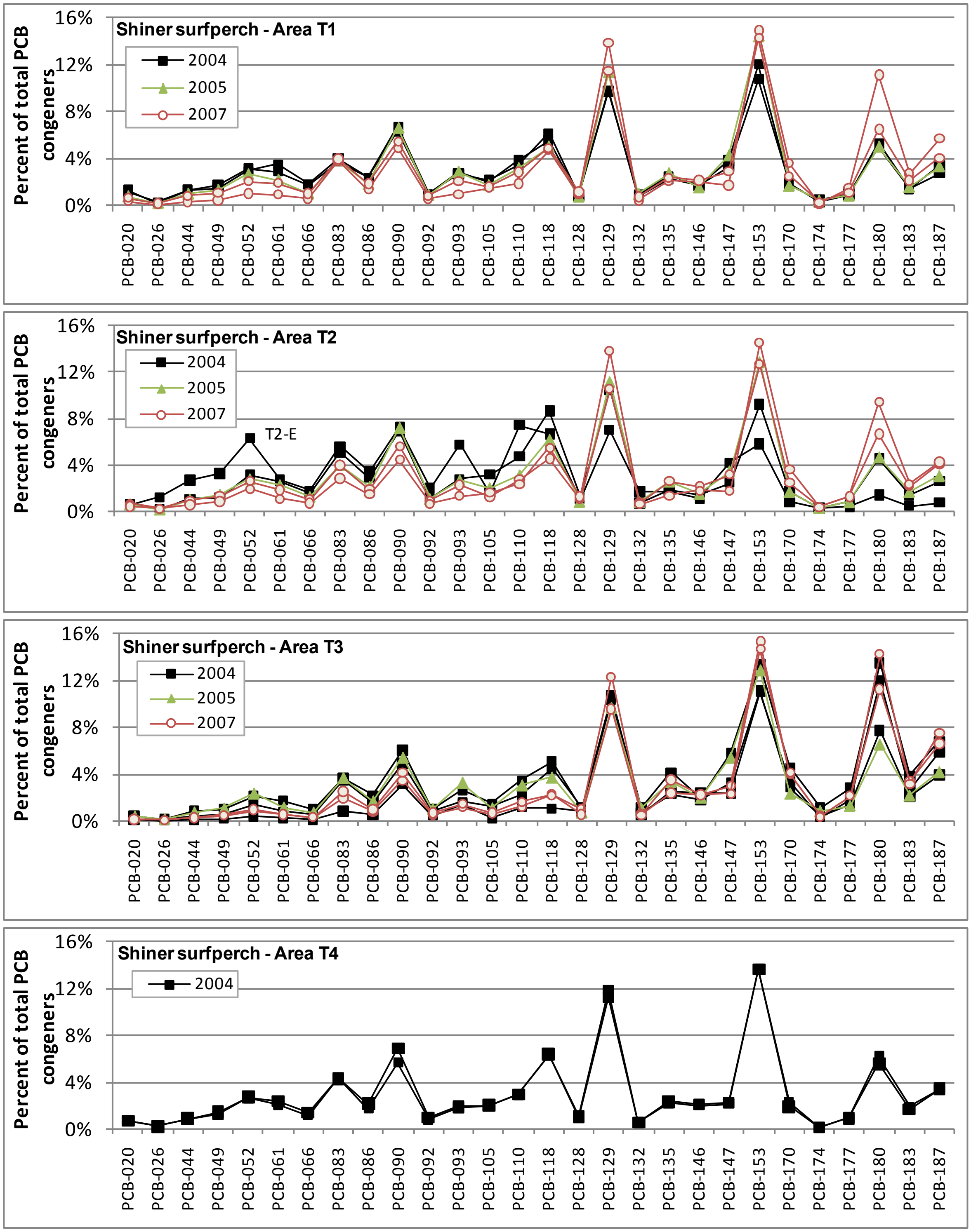
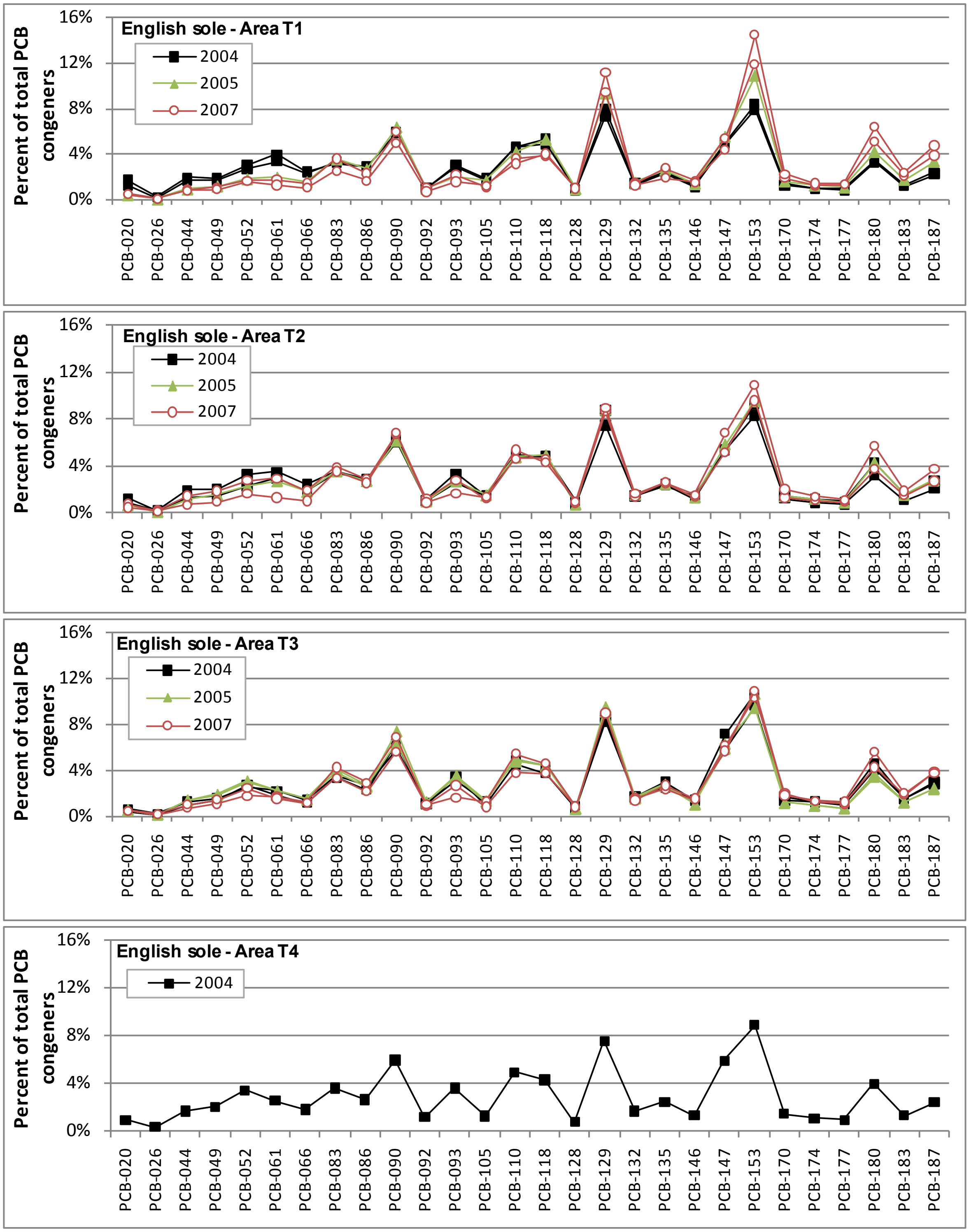
[75] The percentage of total PCB concentrations for each PCB congener within a tissue type was calculated as the mean PCB congener concentration of a particular tissue type divided by the mean total PCB concentration (sum of all PCB congeners) of that tissue type. This analysis was not possible for sediment because sediment samples were not analyzed for all 209 PCB congeners.



Note: Each line represents an individual composite sample.

Figure 4-5. PCB congener patterns in each whole-body composite sample of shiner surfperch collected in 2004, 2005, and 2007 from areas T1, T2, T3, and T4



Note: Each line represents an individual composite sample.

Figure 4-6. PCB congener patterns in each whole-body composite sample of English sole collected in 2004, 2005, and 2007 from Areas T1, T2, T3, and T4

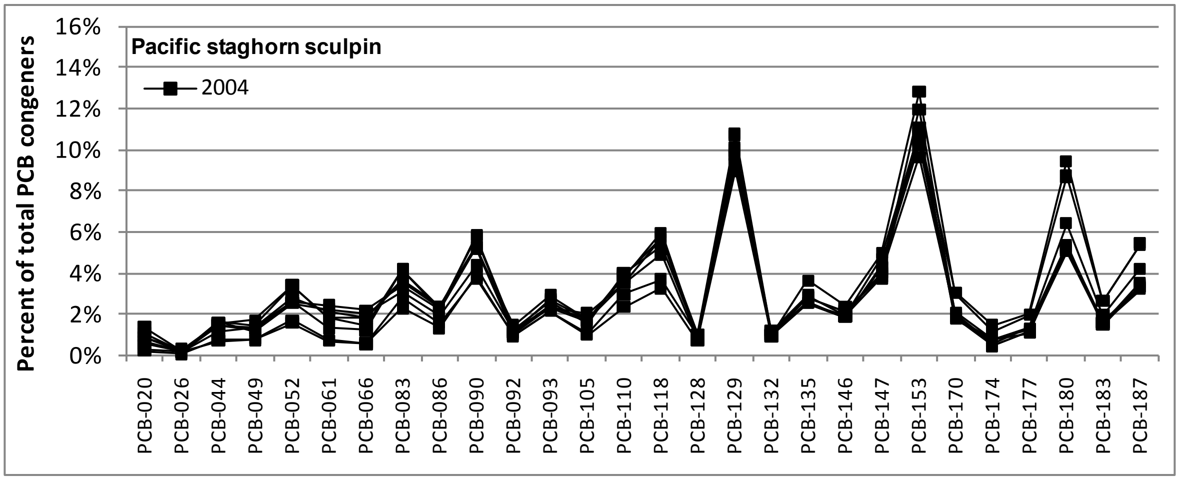
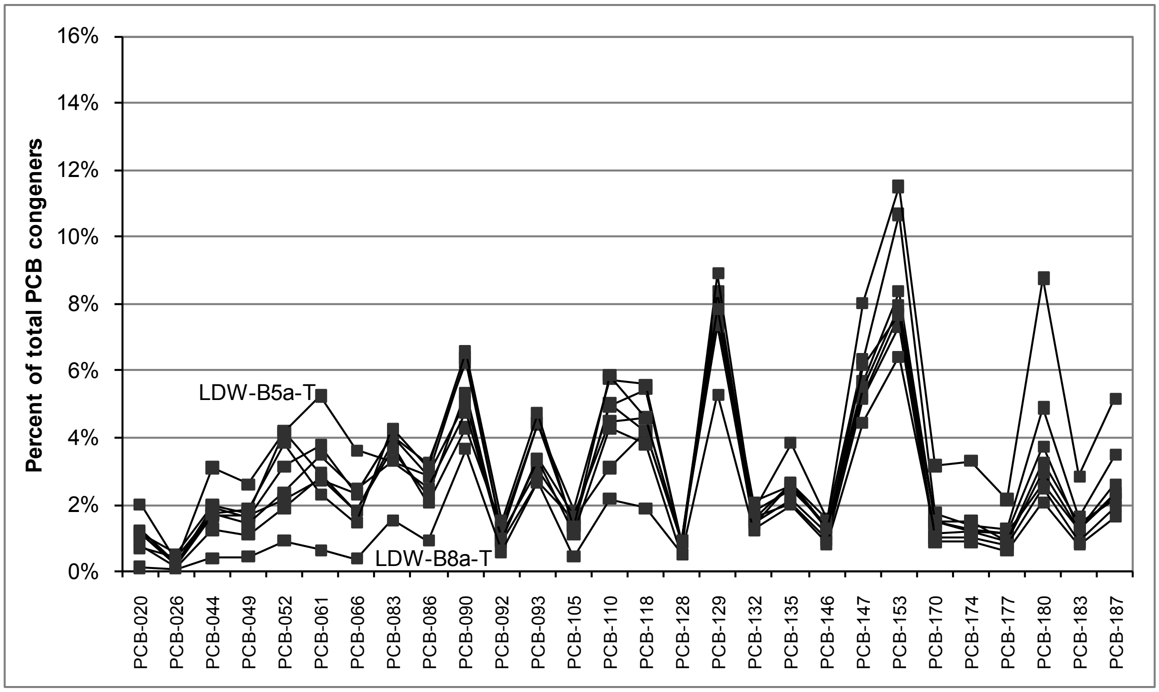


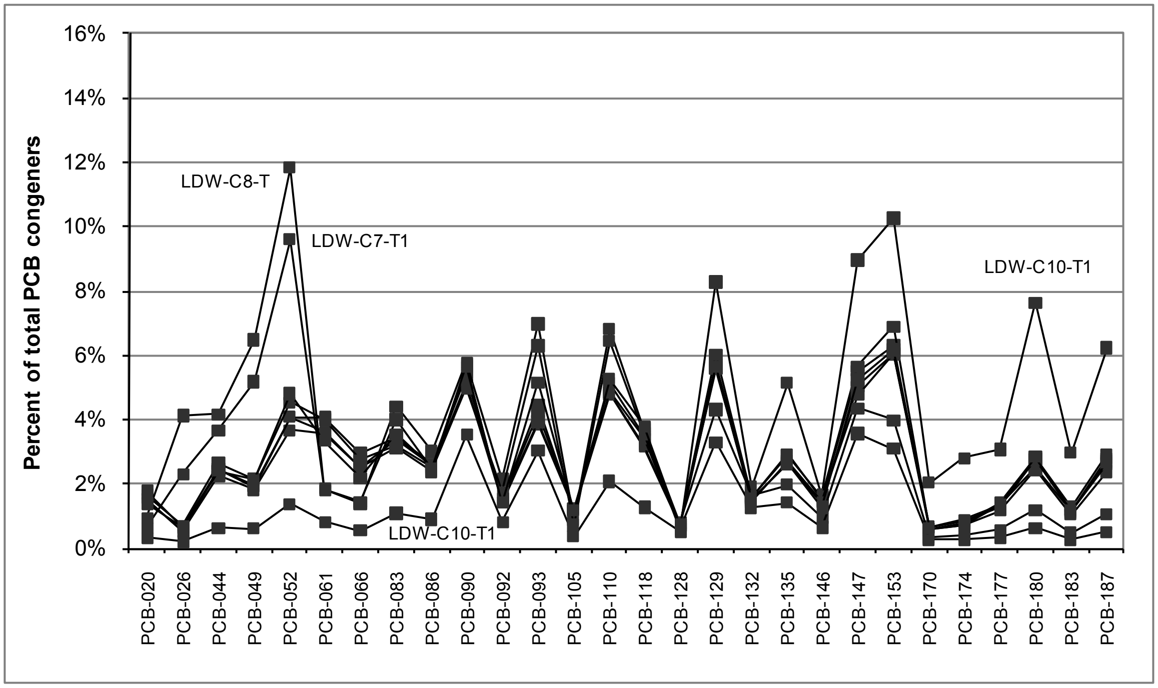
Figure 4-7. PCB congener patterns in each whole-body composite sample of Pacific staghorn sculpin collected in 2004

One benthic invertebrate sample (LDW-B8a from RM 3.5 on the east side of the LDW) had higher relative concentrations of several of the more highly chlorinated PCB congeners (Figure 4-8). Two of the eight clam samples (LDW-C8-T and LDW-C7-T1 from Slip 4) had higher relative concentrations of several less chlorinated PCB congeners, and one clam sample (LDW-C10-T from near RM 3.6 on the west side of the LDW) had relatively high concentrations of more highly chlorinated PCB congeners (Figure 4-9). The influence of the two clam samples from Slip 4 with high contributions of less chlorinated PCB congeners resulted in a mean pattern in clams that differed slightly from the patterns in other tissue types (Figure 4-9). The three clam samples with distinct congener patterns also had the highest total PCB concentrations in clam tissue samples.



Note: Each line represents an individual composite sample.

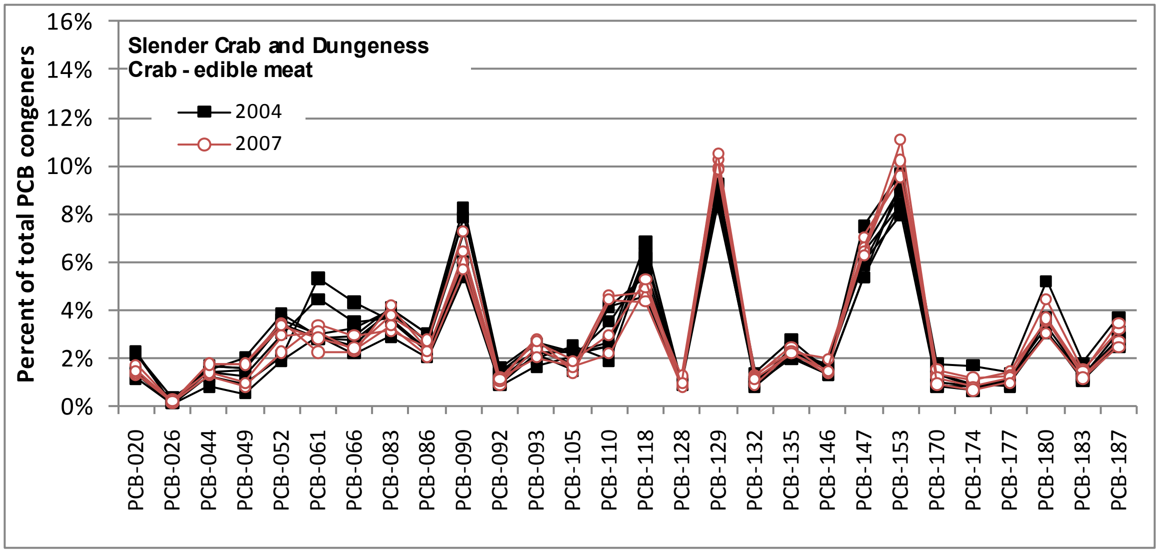
Figure 4-8. **PCB congener patterns in each whole-body composite benthic invertebrate sample collected in 2004**

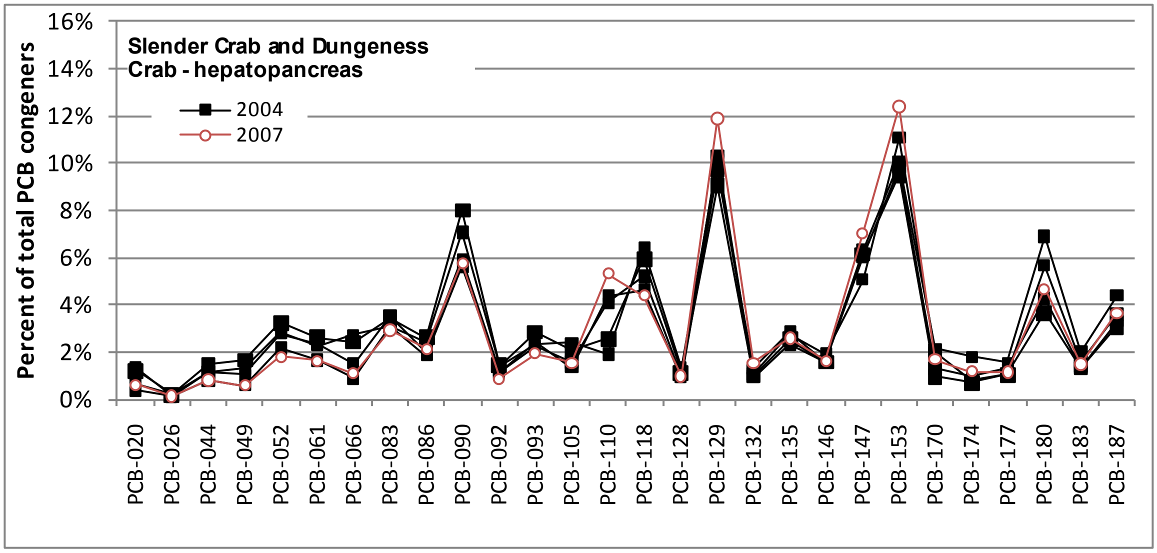


Note: Each line represents an individual composite sample.

Figure 4-9. **PCB congener patterns in each whole-body composite soft-shell clam sample collected in 2004**

The PCB congener patterns for crab edible meat and hepatopancreas samples collected in 2004 and 2007 are presented in Figure 4-10. No discernable differences were observed between the PCB congener patterns in the slender and Dungeness crab samples. The PCB congener pattern in crab tissues collected in 2004 exhibited a slightly higher contribution of the less chlorinated PCB congeners than in 2005, but overall the patterns were similar to one another.





Note: Each line represents an individual composite sample.

Figure 4-10. PCB congener patterns in each edible-meat and hepatopancreas crab composite sample collected in 2004 and 2007

Mean PCB congener patterns in benthic invertebrate, clam, and crab tissue samples are shown relative to one another and to those in fish tissue in Figure 4-11. The mean pattern for whole-body fish samples in 2007 showed a small increase in the contribution from more highly chlorinated PCB congeners relative to the whole-body fish samples from 2004. In general, the PCB congener pattern for the clams had a greater contribution of the less chlorinated PCB congeners than for the other tissue types, potentially reflecting the greater importance of a water exposure route, variation in the ability of different species to metabolize PCBs, or differences in the particular PCB congener mix to which each of these species is exposed.

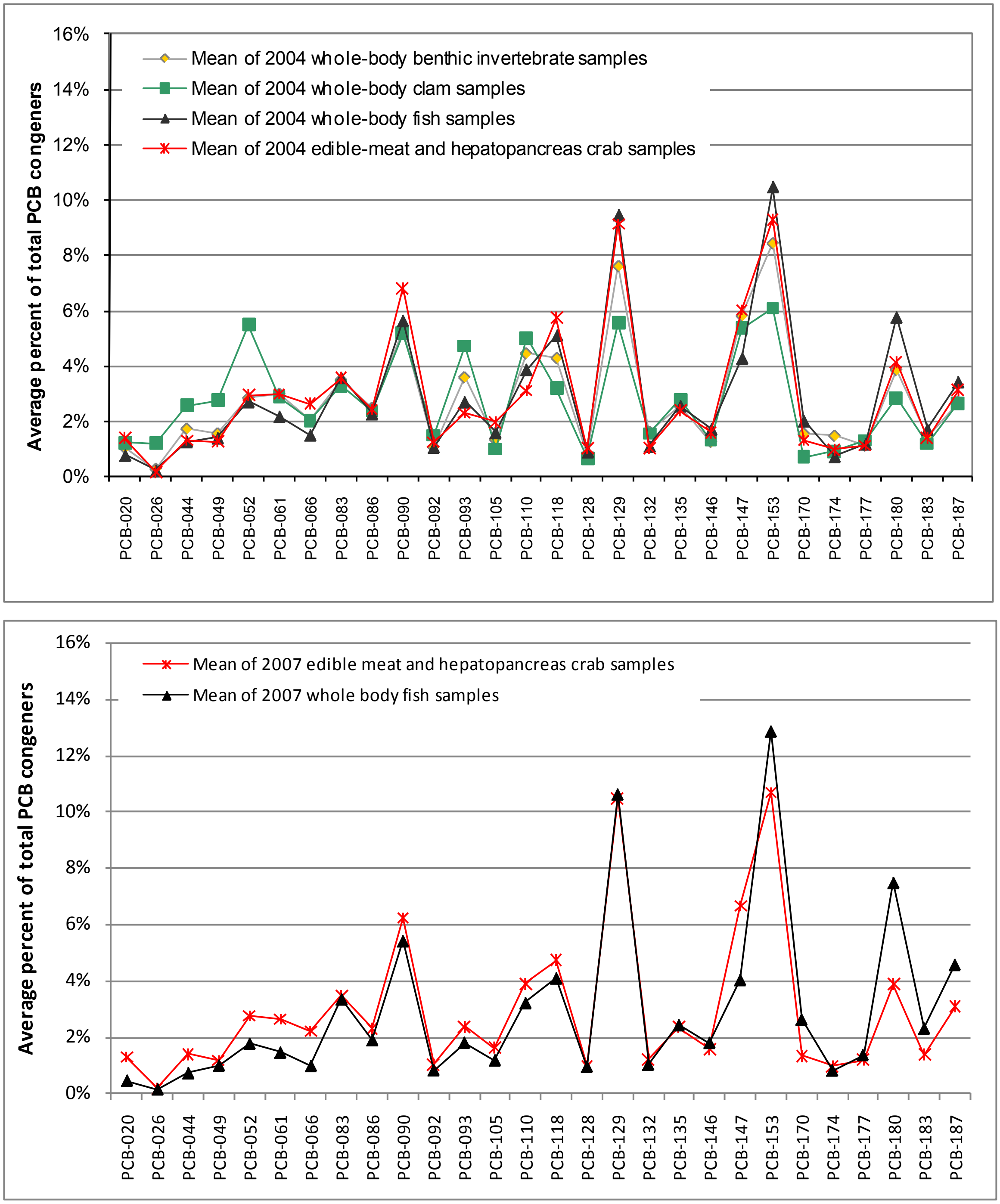


Figure 4-11. Mean PCB congener patterns for benthic invertebrate, clam, fish, and crab samples collected from the LDW in 2004 and 2007

The differences in PCB congener patterns among the shiner surfperch samples collected in 2004 may reflect spatial differences in exposures. This effect is particularly evident in Subareas T2-E, T3-E, and T3-F (Figure 4-4). The three composite samples collected from these subareas in 2004 with distinctly different PCB congener patterns had the three highest total PCB concentrations in fish tissue samples. The variability in the shiner surfperch PCB congener patterns is consistent with the assumption that this species may have a home range smaller than the LDW (Appendix D, Section D.3.2). For English sole, Pacific staghorn sculpin, and crabs, the PCB congener patterns were similar, with no consistent differences among species or collection areas.

PCB Homolog Analysis

EPA conducted an analysis of the relative abundance of the different PCB homolog[76] groups in LDW sediments and tissues based on the 2004 PCB congener data (Kissinger 2006). The homolog concentrations for tissue samples were calculated from the PCB congener concentrations for tissue samples. The homolog concentrations for the sediment samples were calculated from the Aroclor concentrations using the homolog percentages reported for commercial Aroclor mixtures. One sediment PCB homolog pattern was calculated for each of the four fish and crab sampling areas using Thiessen polygons and converting the area-weighted mean Aroclor concentrations into homologs. None of the comparisons have been evaluated statistically because of the small sample size.

The results of this analysis are summarized below.

- English sole, Dungeness crab, and slender crab had similar homolog patterns across the entire site.
- Shiner surfperch and Pacific staghorn sculpin sampled from Area T3 had higher percentages of hexa- and heptachlorobiphenyls than did the shiner surfperch and sculpin collected from Areas T1, T2, and T4.
- Sediment PCB homolog patterns were similar for Areas T1 and T2; tetra-, penta- and hexachlorobiphenyls were the dominant homologs. The homolog pattern for Area T3 was dominated by penta-, hexa-, and heptachlorobiphenyls. The homolog pattern for Area T4 was dominated by tetra- and pentachlorobiphenyls, with substantial contributions from the tri- and hexachlorobiphenyls.

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[76] Homologs are PCB congeners that have the same number of chlorine atoms. These compounds have the same molecular weight and similar physical and chemical properties. The homolog groups are named based on the number of chlorine atoms (e.g., trichlorobiphenyls have three chlorine atoms; hexachlorobiphenyls have six chlorine atoms). The PCB congeners with fewer chlorine atoms have lower molecular weights and tend to be more volatile and more soluble in water.

- The homolog patterns for shiner surfperch and Pacific staghorn sculpin collected in Area T3 were consistent with the homolog pattern calculated for the sediment in this area.

Comparison of Total PCB Aroclor Sums and Total PCB Congener Sums

The sum of the individual PCB congener concentrations in fish and crab tissue samples (Table 4-33) were compared with the sum of the PCB Aroclor concentrations analyzed in the same samples. The sum of PCB congener concentrations is generally believed to provide the most accurate estimate of the total PCB concentration in the samples. Both methods have analytical variability, although Aroclors tend to have greater variability because of uncertainties associated with the identification and quantification of a chemical pattern compared with the identification and quantification of individual PCB congeners. The sums of the PCB congener concentrations in samples collected in 2004 were less than the sums of the Aroclor concentrations (Figure 4-12). In 2005, the PCB congener sums and the Aroclor sums were similar in the three shiner surfperch samples, while the Aroclor sums were less than the PCB congener sums in the three English sole whole-body samples. In 2007, the PCB congener sums were consistently greater than the Aroclor sums. The 2004 samples were analyzed for PCB Aroclors by CAS, and the 2005 and 2007 samples were analyzed for PCB Aroclors by ARI. The change in laboratory may have contributed to the noted differences among years; however, ARI analyzed a subset of the 2004 samples, and the detected PCB Aroclors and sums of PCB Aroclor concentrations were consistent with the CAS results. Overall, the Aroclor sums represented between 30% and 189% of the PCB congener sums for the combined dataset. This range is generally consistent the range of analytical accuracy for the Aroclor analysis (38 to 150%), as specified in the QAPPs (Windward 2004f, 2005g, 2007d).

Table 4-33. Total PCB concentrations (sum of PCB congeners) in composite tissue samples collected from the LDW

		2004				2005				2007			
			Total PCBs (µg/kg ww)				Total PCBs (µg/kg ww)				Total PCBs (µg/kg ww)		
Tissue Type	Area[a]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]
Fish, Whole Body													
English sole	T1	2	1,614 J	2,481 J	2,048	1	2,589 J	2,589 J	2,589	2	774 J	1,165 J	970
	T2	2	2,126 J	2,712 J	2,419	1	3,214 J	3,214 J	3,214	2	1,603 J	1,632 J	1,618
	T3	2	1,419 J	2,457 J	1,938	1	1,433 J	1,433 J	1,433	2	1,032 J	2,928 J	1,980
	T4	1	1,361 J	1,361 J	1,361	0	no data			0	no data		
Shiner surfperch	T1	2	700.1 J	876.6 J	788.4	1	683.1 J	683.1 J	683.1	2	504.1 J	974 J	739
	T2	2	1,055 J	12,228 J	6,642	1	1,047 J	1,047 J	1,047	2	401.6 J	648.3 J	525.0
	T3	3	1,009 J	8,010 J	4,180	1	2,048 J	2,048 J	2,048	2	1,103 J	2,462 J	1,783
	T4	2	532.4 J	770 J	651	0	no data			0	no data		

TISSUE TYPE	AREA[a]	2004				2005				2007			
			TOTAL PCBS (µg/kg ww)				TOTAL PCBS (µg/kg ww)				TOTAL PCBS (µg/kg ww)		
		n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]
Pacific staghorn sculpin	T1	2	532.4 J	668.4 J	600.4	0	no data			0	no data		
	T2	2	481.6 J	496.3 J	489.0								
	T3	2	1,048 J	1,907 J	1,478								
	T4	2	349.6 J	504.9 J	427.3								
Starry flounder	T4	1	458 J	458 J	458	0	no data			0	no data		
Fish, Fillet													
English sole (with skin)	T1	2	857.5 J	1,119.2 J	988.4	0	no data			0	no data		
	T2	2	1,264.6 J	1,269 J	1,266.8								
	T3	2	641.1 J	1,022.9 J	832								
	T4	1	510 J	510 J	510								
Starry flounder (with skin)	T4	1	295.2 J	295.2 J	295.2	0	no data			0	no data		
Pile perch (with skin)	T3	1	192.2 J	192.2 J	192.2	0	no data			0	no data		
Striped perch (with skin)	RM 4.0 – RM 4.1	1	442.3 J	442.3 J	442.3	0	no data			0	no data		
Crab, Edible Meat													
Dungeness crab	T1	1	111 J	111 J	111	0	no data			1	49.45 J	49.45 J	49.45
	T3	1	149.3 J	149.3 J	149.3					1	86.2 J	86.2 J	86.2
	T4	1	148.7 J	148.7 J	148.7					0	no data		
Slender crab	T1	2	174.7 J	186.5 J	180.6	0	no data			1	112 J	112 J	112
	T2	2	129.7 J	180.6 J	155.2					1	86.2 J	86.2 J	86.2
	T3	1	134.3 J	134.3 J	134.3					0	no data		
Crab, Hepatopancreas													
Dungeness crab	T1	0	no data			0	no data			1	612.1 J	612.1 J	612.1 J
	T3	1	3,622 J	3,622 J	3,622					0	no data		
	T4	1	3,618 J	3,618 J	3,618								
Slender crab	T1	1	790.1 J	790.1 J	790.1	0	no data			0	no data		
	T2	1	1,047 J	1,047 J	1,047								
Crab, Whole Body (calc'd)[c]													
Dungeness crab	T1	0	no data			0	no data			1	223.9 JM	223.9 JM	223.9
	T3	1	1,226 JM	1,226 JM	1,226					0	no data		
	T4	1	1,224 JM	1,224 JM	1,224								
Slender crab	T1	2	365.5 JM	373.6 JM	369.6	0	no data			0	no data		
	T2	2	414.06 JM	449.18 JM	431.62								

		2004				2005				2007			
			Total PCBs (µg/kg ww)				Total PCBs (µg/kg ww)				Total PCBs (µg/kg ww)		
Tissue Type	Area[a]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]
Invertebrates, Whole Body													
Benthic Invertebrates	LDW-wide [d]	8	32.13	1,346	393.5	0	no data			0	no data		
Shellfish													
Clams, non-depurated	LDW-wide [e]	8	41.05 J	930 J	222	0	no data			0	no data		

[a] Tissue sampling areas are shown on Maps 4-9 and 4-10.

[b] Mean concentration is the average of detected concentrations. There were no undetected results for total PCBs (as sum of PCB congeners).

[c] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Total PCB concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data).

[d] Benthic invertebrate and clam samples were collected throughout the LDW (Map 4-10).

J – estimated concentration
JM – calculated from an estimated concentration
LDW – Lower Duwamish Waterway
M – calculated concentration
n – number of samples
PCB – polychlorinated biphenyl
RM – river mile
ww – wet weight

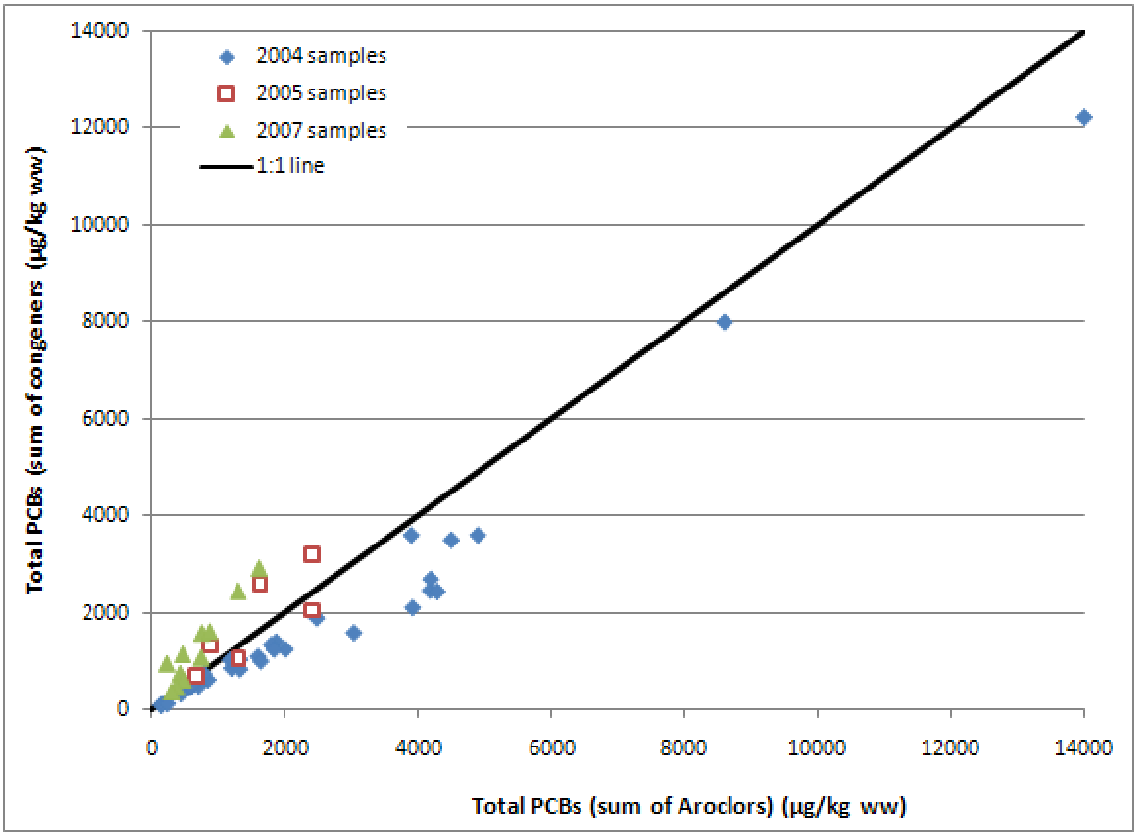


Figure 4-12. Total PCB concentration (sum of PCB congeners) compared with total PCB concentrations (sum of Aroclors) in fish and crab tissue samples

Dioxin Toxic Equivalent Potency

The dioxin toxic equivalent potency of a mixture of dioxin-like PCB congeners can be represented by the ratio of TEQ to total PCBs (sum of PCB congeners) in a sample. The relationship between the TEQ and the total PCB concentration is of interest from a toxicity perspective. Mean potencies calculated for each tissue type collected in 2004, 2005, and 2007 were generally similar to each other (Table 4-34 and Figure 4-13). The consistency of the calculated potencies suggests that the contribution of dioxin-like PCB congeners to the total PCB concentration is relatively consistent over time despite the small shifts in the PCB composition patterns.

Table 4-34. Dioxin toxic equivalent potency in various tissue types collected in 2004, 2005, and 2007

Tissue Type	2004 Samples		2005 Samples		2007 Samples	
	n	Potency Range (mean) (µg TEQ/g PCBs)	n	Potency Range (mean) (µg TEQ/g PCBs)	n	Potency Range (mean) (µg TEQ/g PCBs)
English sole – whole body	7	5.6 – 11.3 (8.9)	3	7.2 – 9.1 (8.2)	6	8.5-9.2 (8.3)
English sole – fillet	7	7.8 – 11.5 (9.8)	0	na	0	na
Pacific staghorn sculpin – whole body	8	5.3 – 13.2 (10.2)	0	na	0	na
Shiner surfperch – whole body	9	4.5 – 17.9 (11.8)	3	7.0 – 12.8 (10.8)	6	5.0-15 (10.9)
Dungeness crab – edible meat	3	9.2 – 12.6 (11.0)	0	na	2	8.8-11 (11.0)
Dungeness crab – hepatopancreas	2	7.5 – 9.3 (8.4)	0	na	1	14

n – number of samples
na – not available
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

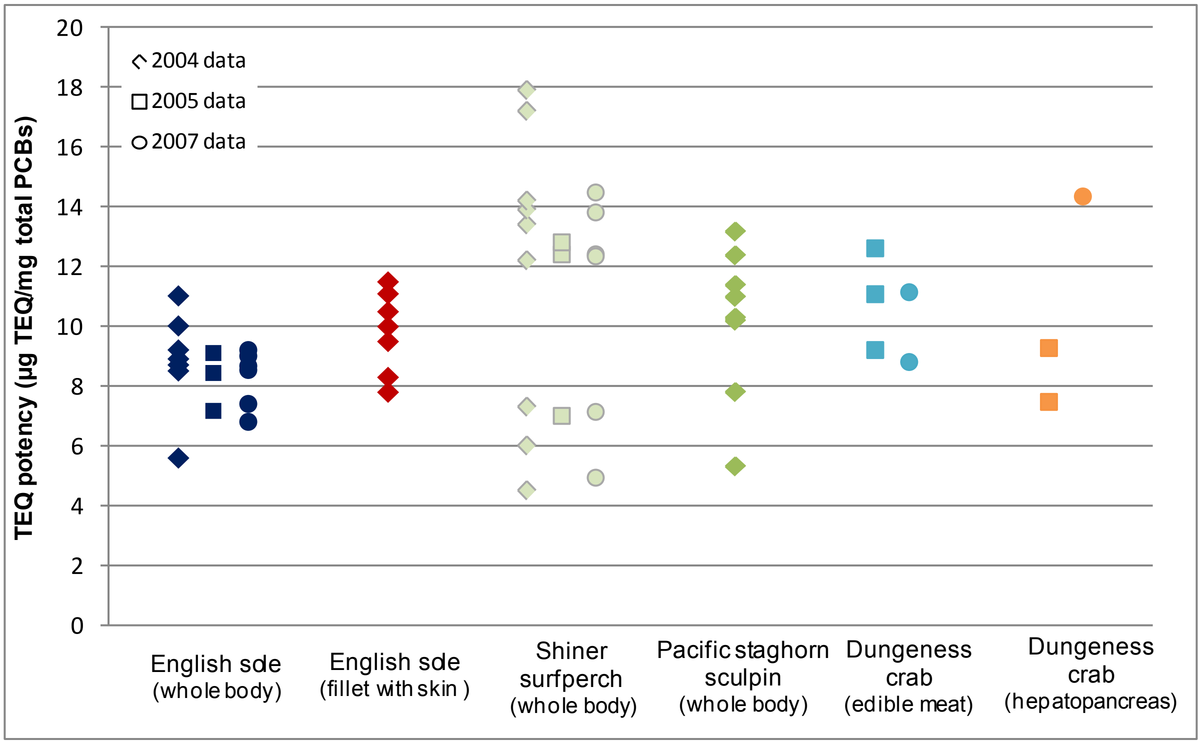


Figure 4-13. Calculated dioxin toxic equivalent potencies for various tissue types

Individual composite samples with the lowest potencies were those with the highest total PCB concentrations. For example, the two shiner surfperch composite samples with the highest total PCB concentrations (sum of PCB congeners) (12,230 and 8,010 µg/kg ww) had the lowest potencies (6.0 and 4.5 µg TEQ/g total PCBs, respectively) because the concentrations of dioxin-like PCB congeners relative to the overall total PCB concentration were relatively low in the mixture. PCB TEQs for the various tissue types are summarized in Table 4-35.

Table 4-35. PCB TEQs in composite tissue samples collected from the LDW

		2004				2005				2007			
			TEQ (µg/kg ww)				TEQ (µg/kg ww)				TEQ (µg/kg ww)		
Tissue Type	Area[a]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]
Fish, Whole Body													
English sole	T1	2	18.2	24.7	21.5	1	23.6	23.6	23.6	2	5.74	10.1	7.92
	T2	2	18.0	23.5	20.8	1	27.1 J	27.1 J	27.1	2	14.5	15.0	14.8
	T3	2	13.0 J	13.7 J	13.4	1	10.3	10.3	10.3	2	7.00	25.0 J	16.0
	T4	1	12.1	12.1	12.1	0	no data			0	no data		
Shiner surfperch	T1	2	9.41 J	15.7	12.6	1	8.49	8.49	8.49	2	7.29	12.1	9.70
	T2	2	18.2	73.0 J	45.6	1	13.4	13.4	13.4	2	5.54 J	8.00	6.77
	T3	3	12.3	36.1	24.7	1	14.4	14.4	14.4	2	7.87	12.2	10.0
	T4	2	7.55 J	10.7 J	9.13	0	no data			0	no data		

Tissue Type	Area[a]	2004				2005				2007			
			TEQ (µg/kg ww)				TEQ (µg/kg ww)				TEQ (µg/kg ww)		
		n	Min	Max	Mean[b]	n	Min	Max	Mean[b]	n	Min	Max	Mean[b]
Pacific staghorn sculpin	T1	2	6.05 J	7.38 J	6.72	0	no data			0	no data		
	T2	2	6.00 J	6.56	6.28								
	T3	2	8.19	10.2	9.20								
	T4	2	3.58 J	5.18 J	4.38								
Starry flounder	T4	1	4.14 J	4.14 J	4.14	0	no data			0	no data		
Fish, Fillet													
English sole (with skin)	T1	2	9.03	12.9	11.0	0	no data			0	no data		
	T2	2	12.7	14.1	13.4								
	T3	2	5.00 J	9.68	7.34								
	T4	1	4.23 J	4.23	4.23								
Starry flounder (with skin)	T4	1	2.9 J	2.9 J	2.9	0	no data			0	no data		
Pile perch (with skin)	T3	1	2.8 J	2.8 J	2.8	0	no data			0	no data		
Striped perch (with skin)	RM 4.0 – RM 4.1	1	6.8 J	6.8 J	6.8	0	no data			0	no data		
Crab, Edible Meat													
Dungeness crab	T1	1	1.4 J	1.4 J	1.4	0	no data			1	0.553	0.553	0.553
	T3	1	1.65	1.65	1.65					1	0.761	0.761	0.761
	T4	1	1.37	1.37	1.37					0	no data		
Slender crab	T1	2	2.52	2.93	2.73	0	no data			1	1.40	1.40	1.40
	T2	2	1.74 J	2.65 J	2.20					1	1.17	1.17	1.17
	T3	1	1.73 J	1.73 J	1.73					0	no data		
Crab, Hepatopancreas													
Dungeness crab	T1	0	no data			0	no data			1	8.80	8.80	8.80
	T3	1	33.6	33.6	33.6					0	no data		
	T4	1	27.1	27.1	27.1								
Slender crab	T1	1	17.1	17.1	17.1	0	no data			0	no data		
	T2	1	12.7	12.7	12.7								
Crab, Whole Body (calc'd)[c]													
Dungeness crab	T1	0	no data			0	no data			1	3.11 M	3.11 M	3.11 M
	T3	1	11.6 M	11.6 M	11.6					0	no data		
	T4	1	9.35 M	9.35 M	9.35								
Slender crab	T1	2	7.04 M	7.32 M	7.18	0	no data			0	no data		
	T2	2	5.13 JM	5.76 JM	5.45								
Invertebrates, Whole Body													
Benthic Invertebrates	LDW-wide[d]	8	0.376 J	8.62 J	3.72	0	no data			0	no data		

		2004				2005				2007			
			TEQ (µg/kg ww)				TEQ (µg/kg ww)				TEQ (µg/kg ww)		
TISSUE TYPE	AREA[a]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]	n	MIN	MAX	MEAN[b]
Shellfish													
Clams, non-depurated	LDW-wide[d]	8	0.440 J	5.65 J	1.48	0	no data			0	no data		

[a] Tissue sampling areas are shown on Maps 4-9 and 4-10.

[b] Mean concentration is the average of detected concentrations. There were no nondetected results for TEQs.

[c] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. TEQs in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data).

[d] Benthic invertebrate and clam samples were collected throughout the LDW (Map 4-10).

J – estimated concentration
JM – calculated from an estimated concentration
M – calculated concentration
n – number of samples
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent
ww – wet weight

TEQs can be estimated from Aroclor concentrations using the calculated potencies of industrial Aroclor mixtures (Rushneck et al. 2004). Figure 4-14 shows the relationship between TEQs calculated using PCB congener concentrations and TEQs estimated from Aroclor concentrations for the same samples. In 2004, the Aroclor-estimated TEQs were similar to the TEQs calculated from the dioxin-like PCB congener concentrations. In 2005 and 2007, the Aroclor-estimated TEQs were less than the PCB congener-based TEQs. The difference may be a result of the fact that Aroclor 1248 was detected in 2004 and not detected in 2005 and 2007. It should be noted that the TEQs used in both the human health and ecological risk assessments were based on PCB congener concentrations. Aroclor-based TEQs were calculated at the request of EPA for the purposes of this comparison.

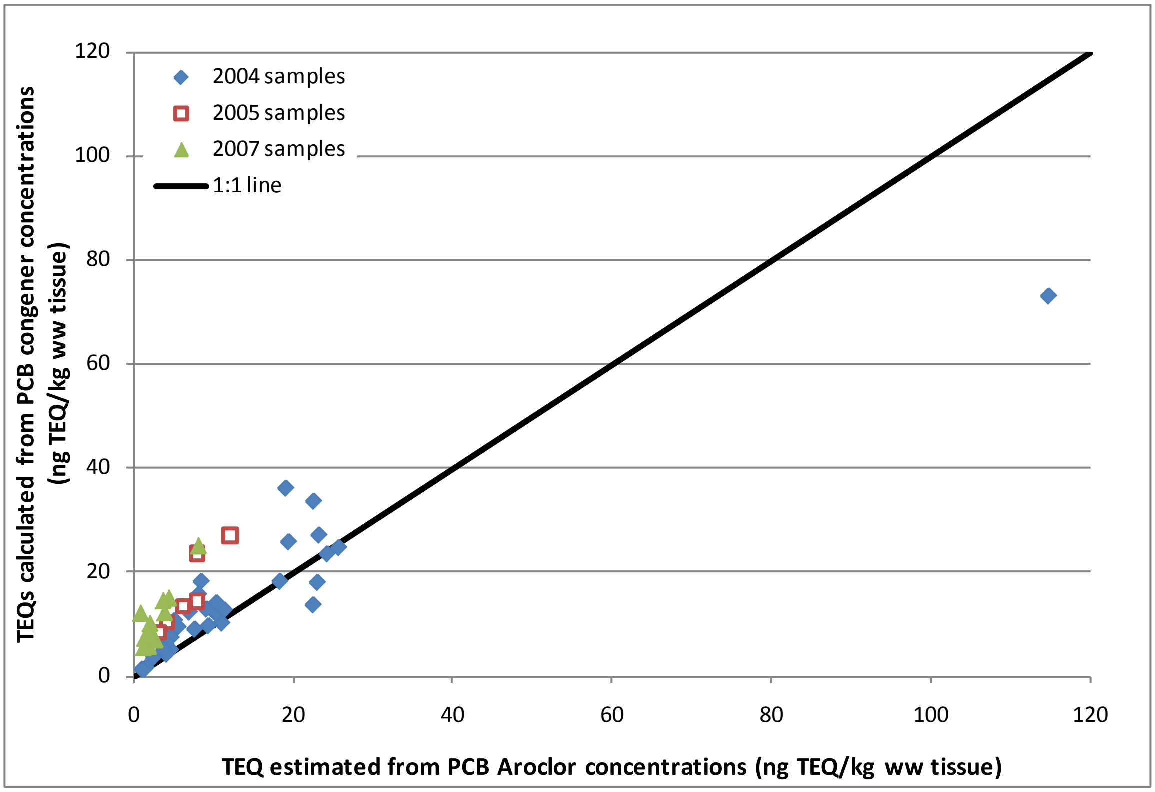


Figure 4-14. TEQs calculated from congener concentrations compared with TEQs estimated from Aroclor concentrations

Temporal Data

Most of the LDW tissue data were collected in 2004, 2005, and 2007 (Table 4-29). The 2004 and 2005 data were collected primarily to support the baseline risk assessments. The 2007 data were collected primarily to provide additional data to assess potential temporal trends. Area T1 was also sampled in 2006 by King County to assist in evaluating trends in T1 where the Duwamish/Diagonal remedial action occurred. Wet-weight and lipid-normalized tissue chemistry data for all of the species in each of the tissue sampling areas and in all sampling areas combined are presented in Appendix E, Section E.4.

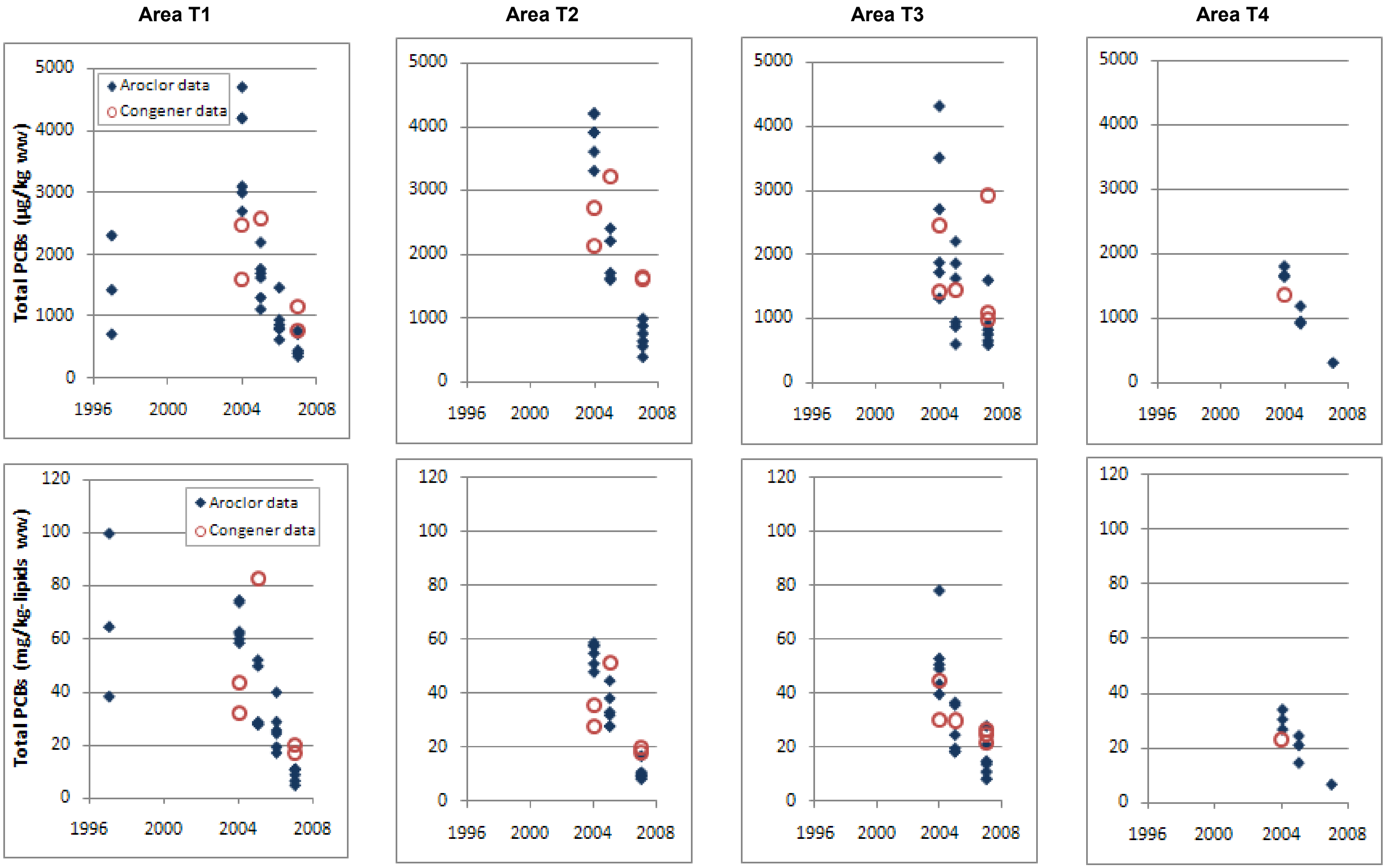
Wet-Weight Data

The highest total PCB concentrations based on Aroclor sums were reported in 2004 for English sole, shiner surfperch, and Dungeness crabs, with decreasing concentrations reported in subsequent years. At the six intertidal locations sampled for clams in both 2004 and 2007, total PCB concentrations in clam tissue were higher at each location in 2004 than in 2007 (Map 4-28). The PCB congener dataset was much smaller than the Aroclor tissue dataset; therefore, temporal trends based on these data are more uncertain.

In general, the total PCB concentrations based on PCB congener sums also appeared to decrease in English sole tissue in Area T1 and to some degree in Area T2; the data for Area T3 were less clear (Figure 4-15). The total PCB concentrations (PCB congener sums) in shiner surfperch tissue did not indicate a downward or upward trend in any of the areas, with the possible exception of a potential downward trend in Area T2 (Figure 4-16). Even fewer PCB congener data are available for crab tissues. However, every total PCB concentration (PCB congener sums) in 2007 was less than the corresponding concentration in 2004 (corresponding data are available for Dungeness and slender crab edible meat; Figure 4-17 and Table 4-33), potentially supporting the downward trend based on the Aroclor data.

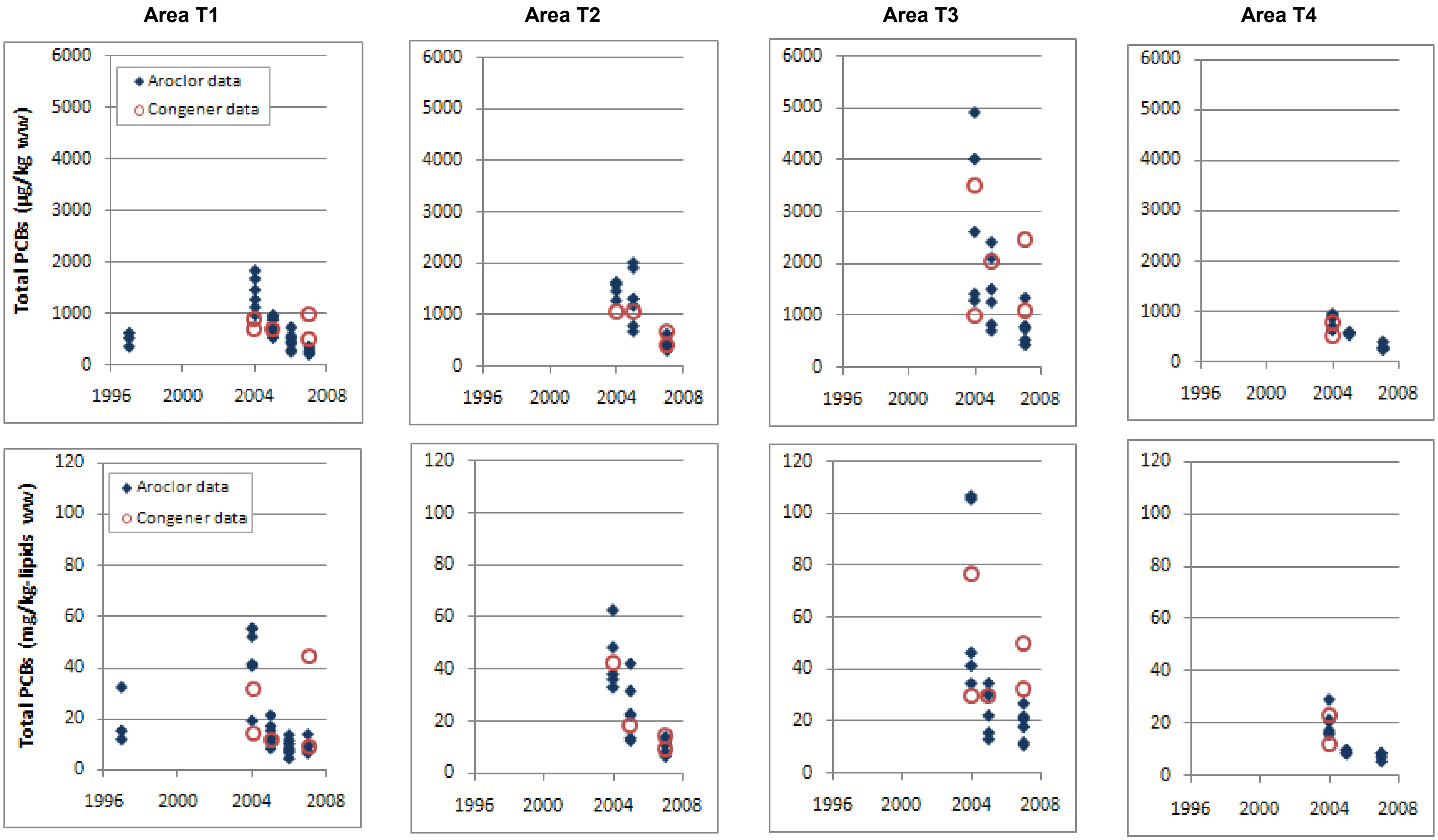
There are several uncertainties associated with these trends. The trends based on the PCB congener data are highly uncertain because of the small sample size of the dataset (i.e., the higher the variability, the more uncertain the interpretation of the small dataset). Regarding the Aroclor data, PCB Aroclors were analyzed by CAS in 2004 and by ARI in 2005, 2006, and 2007. In addition, the PCB congener pattern changed over time (2004 vs. 2005, 2006, and 2007), which may have affected the quantification of Aroclors.

The Aroclor wet-weight tissue data as a whole suggest total PCB concentrations in 2004 were higher than those in more recent years; total PCB concentrations appear to be decreasing based on Aroclor data. Trends based on the limited PCB congener data are more uncertain. The PCB congener data may suggest total PCB concentrations in English sole collected from Area T1 and possibly from Area T2 are decreasing, although the sample size is small. Total PCB concentrations (congener sums) in crabs collected from Areas T1 and T3 were also consistently lower in 2007 than 2004. The PCB congener data suggest temporal trends in total PCB concentrations in shiner surfperch are uncertain.



Note: English sole whole-body samples from 1997 consisted of tissue remaining after subsamples of fillets were removed. In addition, livers were removed from some fish in these composite samples. These data are not included in the RI dataset because of these uncertainties, but are shown here for informational purposes.

Figure 4-15. Temporal comparison of total PCB concentrations based on the sum of Aroclors and the sum of individual PCB congeners in English sole whole-body tissue by tissue sampling area



Note: The maximum concentrations in Areas T2 and T3 (both from 2004) are not shown. In Area T2, the maximum concentration was 18,400 µg/kg dw (329 mg/kg lipids) (Aroclor sum) or 12,228 µg/kg (218 mg/kg lipids) (congener sum). In Area T3, the maximum concentration was 8,800 µg/kg dw (284 mg/kg lipids) (Aroclor sum) or 8,010 µg/kg (258 mg/kg lipids) (congener sum).

Figure 4-16. Temporal comparison of total PCB concentrations based on the sum of Aroclors and the sum of individual PCB congeners in shiner surfperch whole-body tissue by tissue sampling area

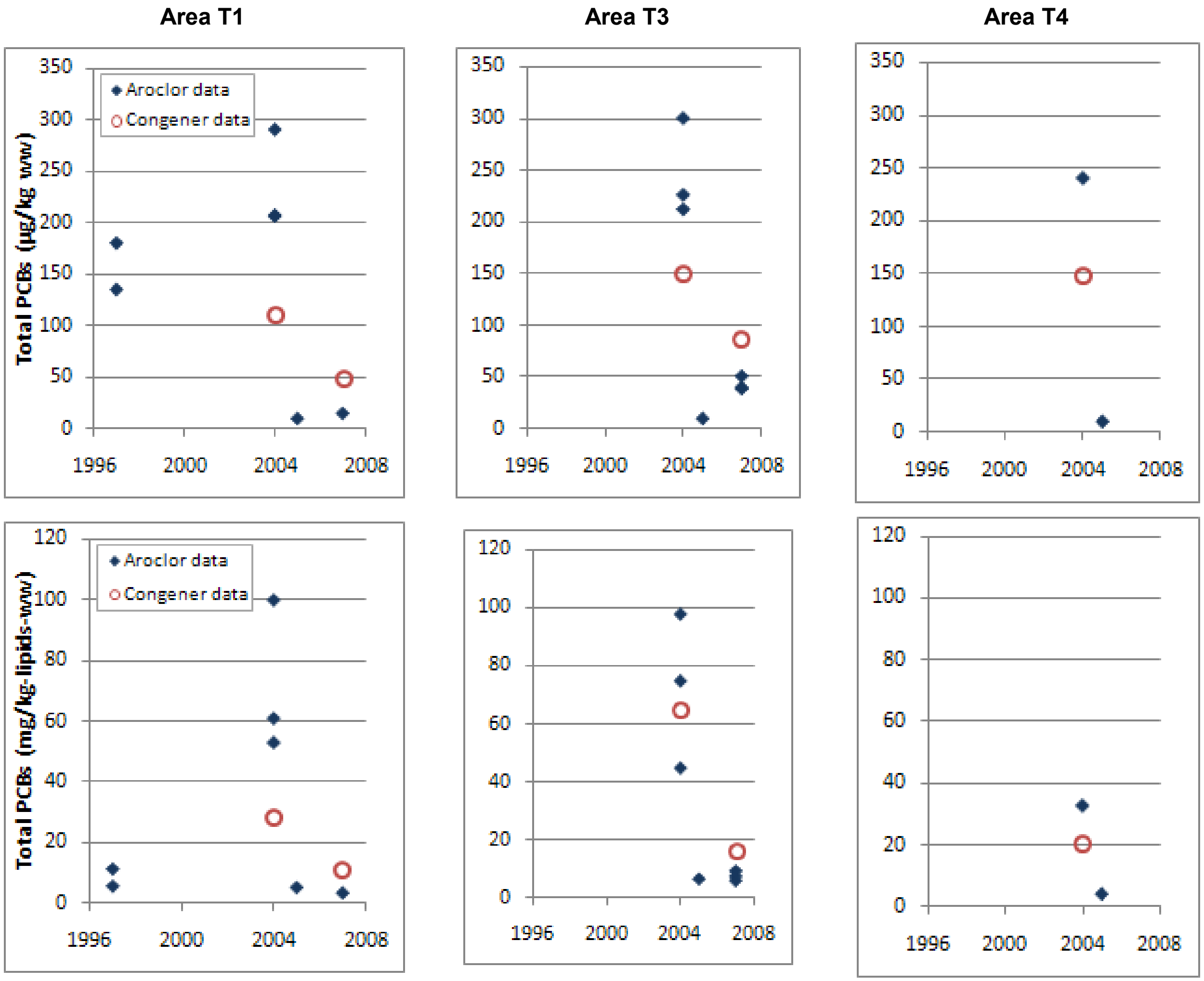


Figure 4-17. Temporal comparison of total PCB concentrations based on the sum of Aroclors and the sum of individual PCB congeners in Dungeness crab edible meat tissue by tissue sampling area (no Dungeness crabs were collected in Area T2)

Lipid-Normalized Data

The highest lipid-normalized total PCB concentrations based on Aroclor sums were reported in 1997 and 2004 for English sole (Figure 4-15) and in 2004 for shiner surfperch (Figure 4-16) and Dungeness crabs (Figure 4-17), with decreasing concentrations reported in subsequent years. Thus, similar to the wet-weight data, the lipid-normalized Aroclor data suggest that total PCB concentrations are decreasing over time.

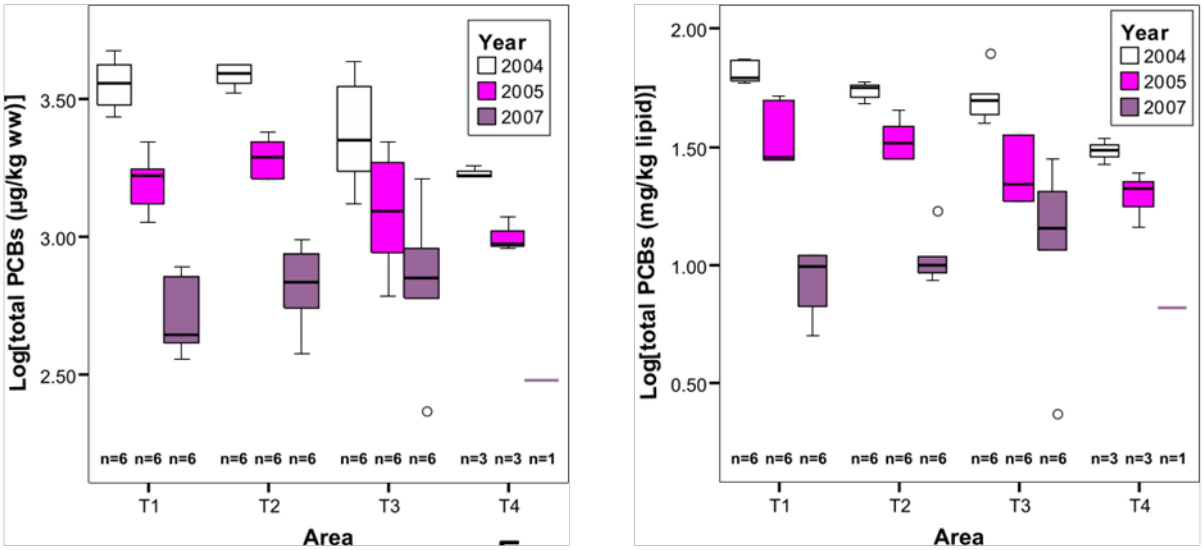
As discussed above for wet-weight data, the PCB congener dataset was much smaller than the Aroclor tissue dataset; therefore, temporal trends based on these data are more uncertain. PCB congener data are available in Area T4 only for 2004 and are not available for crabs in any year in Area T2. Based on the small dataset, there are potential downward trends in lipid-normalized total PCB concentrations in Area T3 for English sole, Area T2 for shiner surfperch, and Areas T1 and T3 for Dungeness crab (Figures 4-15, 4-16, and 4-17, respectively).

Spatial Trends

To evaluate whether differences were statistically significant among mean total PCB concentrations (sum of Aroclors in µg/kg ww) in the four sampling areas in whole-body English sole, shiner surfperch, and Pacific staghorn sculpin samples, analysis of variance (ANOVA) and *post hoc* tests were conducted. Two-way ANOVAs were conducted with the factors of year (2004, 2005, and 2007), area (T1 through T4), and year-by-area interaction. These tests were followed by *post hoc* Tukey honestly significant difference multiple comparisons to identify differences between areas (averaged over all years) and one-way ANOVA testing for effects of area within a year (Sidak's correction used to correct for multiple testing) to identify differences between particular years and to help evaluate the effects of small sample size in Area T4. Data were log-transformed to help homogenize variances in order to better meet the assumptions of the ANOVA and were also rankit-transformed to make the variables normally distributed and the analyses "non-parametric"(Conover 1980). Similar analyses were also conducted for lipid-normalized total PCB concentrations (sum of Aroclors in mg/kg lipid). Sufficient data were available for 2004, 2005, and 2007 for English sole (though sample sizes were small in area T4) and shiner surfperch and for only 2004 for Pacific staghorn sculpin. Data sufficient for statistical analyses were not available for crabs. Total PCB concentrations in tissues are shown for the four sampling areas on Map 4-28, with sediment SWACs included for context (see Appendix D.6 for correlations evaluated as part of the FWM). Results of the statistical analyses for each of the three fish species are presented below. Complete tables of p values for all analyses conducted are presented in Appendix E, Section E.5.

Whole-Body English Sole Wet-Weight Data

Means of wet-weight total PCB concentrations were lower in Area T4 than in all other areas in all years and averaged over all years (p = 0.05, Figures 4-15 and 4-18). The two-way ANOVAs for log- and rankit-transformed total PCBs in whole-body English sole indicated that at least one area had a mean total PCB concentration (averaged over all years) that differed from at least one other area (significant area effect but no interaction between year and area for all variables[Appendix E5, Table E.5-1]). Tukey *post hoc* tests identified significant differences between Areas T4 and T2 (averaged over all years) for log-transformed wet-weight total PCB concentrations (see Appendix E, Tables E.5-2 and E.5-3, for p values).



Note: The boxes indicate the 25th and 75th percentiles within which 50% of the data lie. The length of this box indicates the interquartile range (IQR). The line in the center of the box is the 50th percentile (median). The whiskers of the box plot indicate the maximum and minimum values from the dataset that are within 1.5 IQRs of the 25th and 75th percentile values. Outliers are indicated by a circle and have a value between 1.5 and 3 of the IQR from the upper or lower edge of the box (75th or 25th percentile).

Figure 4-18. Box plots of log(10)-transformed wet-weight and lipid-normalized total PCB concentrations (Aroclor sum) in English sole collected in 2004, 2005, and 2007

Based on within-year ANOVAs (using Sidak's correction for testing of multiple years), area means for total PCB concentrations were not significantly different in 2005 and 2007 (Table 4-36). In 2004, significant differences were identified only between the area with the lowest mean (Area T4) and the areas with the highest or second highest mean (Areas T1 or T2) (see Appendix E, Table E.5-5, for p values).

Table 4-36. Results of *post hoc* one-way ANOVAs for equality of area mean total PCB concentrations (Aroclor sum) in English sole within year

Type of Data		Year			All Years[a]
		2004	2005	2007	
Log[total PCBs (µg/kg ww)]	highest	T2	T2	T3	T2
		T1	T1	T2	T1
		T3	T3	T1	T3
	lowest	T4	T4	T4	T4
Log[total PCBs (mg/kg lipid)]	highest	T1	T1	T3	T1
		T2	T2	T2	T2
		T3	T3	T1	T3
	lowest	T4	T4	T4	T4

Note: Test was conducted using Sidak's correction for multiple comparisons. Areas are listed from highest to lowest mean concentration. Areas connected with vertical lines are not significantly different.

[a] Differences among all year means are based on Tukey *post hoc* tests following two-way ANOVA for area, year, and area-by-year interaction.

PCB – polychlorinated biphenyl
ww – wet weight

Whole-Body English Sole Lipid-Normalized Data

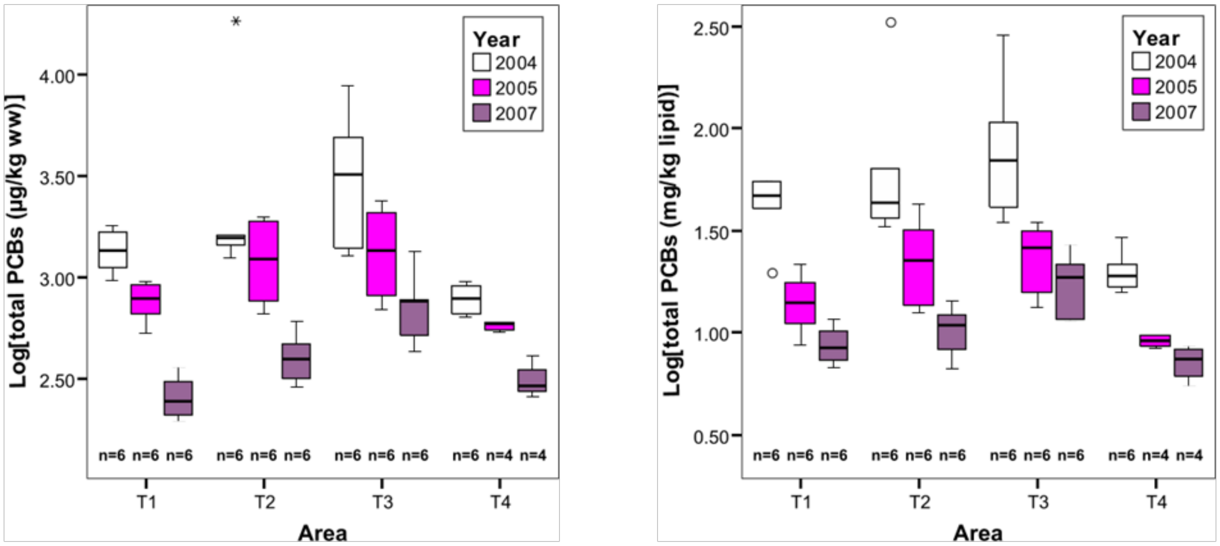
Means of lipid-normalized total PCB concentrations were lower in Area T4 than in all other areas in all years and averaged over all years (Figures 4-15 and 4-18). The two-way ANOVA indicated that for lipid-normalized total PCBs, at least one area had a mean concentration (averaged over all years) that differed from at least one other area (area effect of the two-way ANOVA was significant; $p = 0.022$). Tukey *post hoc* tests identified no significant differences among areas in log-transformed lipid-normalized total PCB concentrations (see Appendix E, Section E.5, for p values).

Based on individual, within-year ANOVAs (using Sidak's correction for testing of multiple years), area means were not significantly different in 2005 and 2007 (Table 4-36). In 2004, significant differences were identified only between the area with the lowest mean (Area T4) and the area with the highest mean (Area T1) (see Appendix E, Section E.5, for p values).

Whole-Body Shiner Surfperch Wet-Weight Data

Means of wet-weight total PCB concentrations were higher in Areas T2 and T3 and lower in Areas T1 and T4 in all years and averaged over all years (Figure 4-19). The two-way ANOVA indicated that for wet-weight total PCBs, at least one area had a mean concentration (averaged over all years) that differed from at least one other area (area effect of the two-way ANOVA was significant; $p < 0.0005$). Tukey *post hoc* tests identified no significant differences (averaged over all years) between log-transformed wet-weight total PCB concentrations in Areas T4 and T1 or between concentrations in Areas T2 and T3 but indicated that the means in the two groups (T4, T1 and T3, T2) were significantly different (Table 4-37; see Appendix E, Section E.5, for p values).

Based on one-way ANOVA testing for differences among areas within years, log-transformed wet-weight total PCB concentrations did not differ among areas in 2005 but were lower in Areas T1 and T4 than in Areas T2 and T3 in 2004 and 2007 (Table 4-37; see Appendix E, Section E.5, for p values).



Note: The boxes indicate the 25th and 75th percentiles within which 50% of the data lie. The length of this box indicates the interquartile range (IQR). The line in the center of the box is the 50th percentile (median). The whiskers of the box plot indicate the maximum and minimum values from the dataset that are within 1.5 IQRs of the 25th and 75th percentile values. Outliers are indicated by a circle and have a value between 1.5 and 3 of the IQRs from the upper or lower edge of the box (75th or 25th percentile). Extreme values are indicated by an asterisk and have a value > 3 IQRs from the 75th or 25th percentile.

Figure 4-19. Box plots of log(10)-transformed wet-weight and lipid-normalized total PCB concentrations (Aroclor sum) in shiner surfperch collected in 2004, 2005, and 2007

Table 4-37. Results of *post hoc* one-way ANOVAs for equality of area mean total PCB concentrations (Aroclor sum) in shiner surfperch within year

Type of Data		Year			All Years[a]
		2004	2005	2007	
Log[total PCBs (µg/kg ww)]	Highest Lowest	T2 T 3 T1 T 4	T3 T 2 T 1 T 4	T3 T 2 T4 T 1	T3 T 2 T1 T 4
Log[total PCBs (mg/kg lipid)]	Highest Lowest	T3 T 2 T 1 T 4	T3 T 2 T 1 T 4	T3 T 2 T 1 T 4	T3 T2 T1 T 4

Note: Test was conducted using Sidak's correction for multiple comparisons. Areas are listed from highest to lowest mean concentration. Areas connected with vertical lines are not significantly different.

[a] Differences among all year means are based on Tukey *post hoc* tests following two-way ANOVA for area, year, and area-by-year interaction.

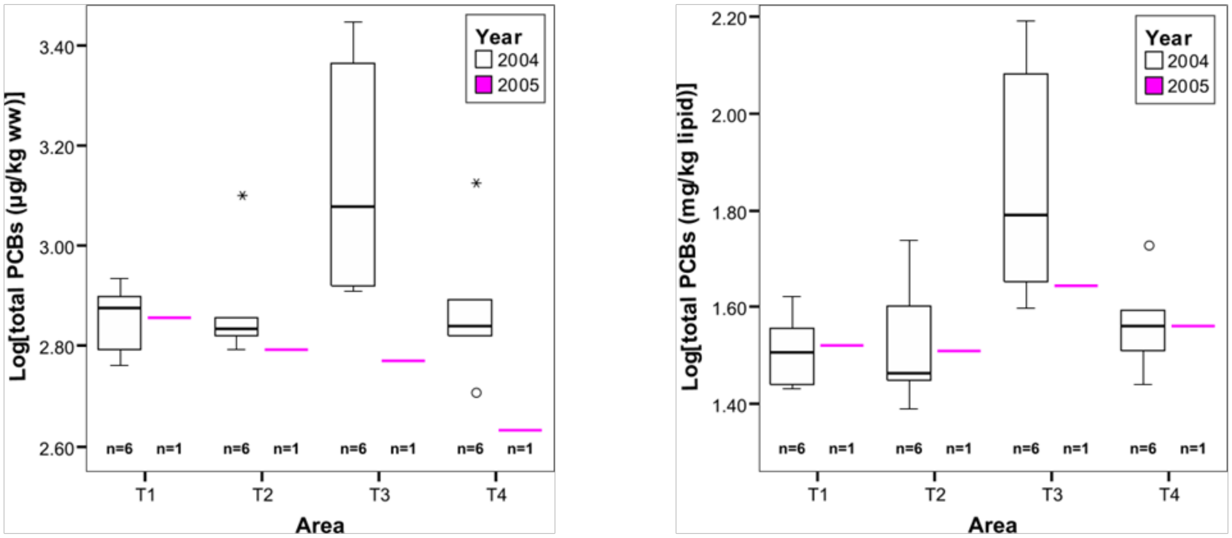
PCB – polychlorinated biphenyl

Whole-Body Shiner Surfperch Lipid-Normalized Data

Means of lipid-normalized total PCB concentrations were higher in Areas T2 and T3 and lower in Areas T1 and T4 in all years and averaged over all years (Figure 4-19). The two-way ANOVA indicated that for lipid-normalized total PCBs, at least one area had a mean concentration (averaged over all years) that differed from at least one other area (area effect of the two-way ANOVA was significant; $p < 0.0005$). Tukey *post hoc* tests for differences among all year means identified significant differences between Area T4 and Areas T3 and T2 based on log-transformed lipid-normalized total PCB concentrations (Table 4-37; see Appendix E, Section E.5, for p values). Based on one-way ANOVA testing for differences among areas within years, differences among areas were not significant in any year (Table 4-37; see Appendix E, Section E.5, for p values).

Whole-Body Pacific Staghorn Sculpin Wet-Weight and Lipid-Normalized Data

For Pacific staghorn sculpin, there were sufficient data in each of the four sampling areas to conduct statistical analyses of differences among the areas in 2004 but not in 2005 (six samples were collected in each area in 2004, and one sample was collected in each area in 2005). Box plots of log-transformed wet-weight and lipid-normalized total PCB concentrations in Pacific staghorn sculpin are shown in Figure 4-20. The one-way ANOVA using 2004 log-transformed total PCB concentrations identified a significant difference among area means ($p = 0.01$). *Post hoc* Tukey tests identified significant differences in the means of log-transformed total PCB concentrations between Area T3 and all other areas (p values ranged from 0.02 to 0.03). Conclusions were the same using log-transformed lipid-normalized total PCB concentrations.



Note: The boxes indicate the 25th and 75th percentiles within which 50% of the data lie. The length of this box indicates the interquartile range (IQR). The line in the center of the box is the 50th percentile (median). The whiskers of the box plot indicate the maximum and minimum values from the dataset that are within 1.5 IQRs of the 25th and 75th percentile values. Outliers are indicated by a circle and have a value between 1.5 and 3 of the IQR from the upper or lower edge of the box (75th or 25th percentile). Extreme values are indicated by an asterisk and have a value > 3 times the IQR from the 75th or 25th percentile.

Figure 4-20. Box plots of log(10)-transformed wet-weight and lipid-normalized total PCB concentrations in Pacific staghorn sculpin in 2004 and 2005

Tissue and Sediment Relationships

Relationships between total PCB concentrations in fish or crab tissue (ww) and surface sediment (dw) were evaluated as part of the FWM and are described in detail in Appendix D (Section D.3). Regression analyses to assess the relationships were conducted using data for the 24 subareas for shiner surfperch and Pacific staghorn sculpin, the two species believed to have smaller home ranges, and data from the four larger tissue sampling areas for English sole and crabs (see Map 4-9 for area locations). Significant relationships between tissue and surface sediment on a subarea basis were identified for shiner surfperch and Pacific staghorn sculpin using 2004 data; total PCB concentrations in surface sediment explained more than 50% of the variance in concentrations in tissue. Using 2005 data for shiner surfperch, the relationship was significant but less strong, explaining 29% of the variance. Sufficient data were not available to evaluate the relationship for Pacific staghorn sculpin in 2005. For English sole and crabs, regression relationships were not significant on an area basis using either 2004 or 2005 data.

The relationship between total PCB concentrations in benthic invertebrate tissue (ww) and co-located surface sediment (dw) was evaluated as part of the ERA (Appendix A; Attachment 11). The relationship was significant, with total PCB concentrations in surface sediment explaining 67% of the variance in tissue. The relationship between total PCB concentrations in clams (ww) and co-located sediment (dw) was evaluated as part of the FWM (Appendix D; Section D.7.3). The sediment variable explained 80% of the variance in tissue concentrations ($R^2 = 0.80$), and the regression was significant.

4.2.3.5 Surface water

Concentrations of PCB congeners in surface waters are available from two different sampling methods and events. In 1997, King County used SPMDs to estimate PCB concentrations in surface waters of the LDW as part of the King County WQA. These data are not summarized in the RI because only a subset (17 of 209) PCB congeners were estimated using the SPMDs. It is likely the total PCB concentrations were underestimated because the totals did not include all PCB congeners.[77] In 2005, King County collected whole water samples that were analyzed by Axys for all 209 congeners (Table 4-38) (Mickelson and Williston 2006). The surface water samples were collected from two locations in the LDW: RM 0.0 at the southern end of Harbor Island (LTKE03) and RM 3.3 at the South Park Bridge (LTUM03). Each of these locations was sampled during two dry-weather periods (August and September) and two wet-weather periods (November and December). All 209 PCB congeners were analyzed in unfiltered water samples using high-resolution methods. Total PCB concentrations were calculated as the sum of detected PCB congeners.

Table 4-38. Summary of surface water data for total PCBs (as sum of PCB congeners)

		Total PCB Concentration and Salinity by Date[a]							
		8/22/2005 (277 cfs)[b]		9/26/2005 (378 cfs)[b]		11/28/2005 (1,060 cfs)[b]		12/19/2005 (550 cfs)[b]	
Location ID	Collection Depth	Total PCBs (µg/L)	Salinity (PSS)	Total PCBs (µg/L)	Salinity (PSS)	Total PCBs (µg/L)	Salinity (PSS)	Total PCBs (µg/L)	Salinity (PSS)
LTKE03 (RM 0.0)	1 m below the surface	0.001796	22.984	0.001024	25.174	0.000591	13.388	0.001947 J	25.987
	1 m above the bottom	0.001814	28.273	nc[c]	30.266	0.00025	30.118	0.000599	29.995
LTUM03 (RM 3.3)	1 m below the surface	0.001592 J	16.523	0.001452 J	17.133	0.000398	9.929	0.001122	9.423
	1 m above the bottom	0.003211	26.043	0.001883 J	29.402	0.000132	20.362	0.001341	27.775

[a] Total PCB concentration represents the sum of detected PCB congener concentrations. RLs for non-detects were not included in the calculation. Laboratory duplicate and field replicate samples were averaged using the data management rules presented in Appendix E, Section E.3. Detected results for select PCB congeners that were within 5 times the detected concentration in the associated method blank sample were qualified as non-detected at elevated RLs. Alternate data management procedures and data validation criteria were used to

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[77] Total PCB concentrations calculated from SPMD data ranged from 0.000092 to 0.000193 µg/L based on the sum of the 17 estimated PCB congener concentrations. Total PCB concentrations were also calculated using Aroclor data collected with SPMDs (ranging from 0.000161 to 0.000320 µg/L), but these data are highly uncertain because Aroclor-specific partitioning coefficients were not available (King County 1999e). These concentrations were generally lower than total PCB concentrations in whole-water samples collected from the LDW in 2005.

calculate total PCB concentrations in the King County technical memorandum that presented these data (Mickelson and Williston 2006).

[b] Daily mean discharge flow rate in the Green River at USGS Gauge 12113000 at Auburn.

[c] A number of PCB congener results were rejected because method performance criteria were not met during analysis; therefore, total PCB concentrations were not calculated.

cfs – cubic feet per second
ID – identification
J – estimated concentration
nc – not calculated
PCB – polychlorinated biphenyl
PSS – practical salinity scale
RL – reporting limit

Total PCB concentrations (sum of PCB congener concentrations) in water at both depths in both locations were lowest in November. Flow, represented as the daily mean discharge from USGS Gauge 12113000 at Auburn, was also highest in November (1,060 cfs) than in August, September, and December (277, 378, and 550 cfs, respectively).

The highest flow rate was associated with the lowest total PCB concentration at each LDW location and depth, and the lowest flow rate was associated with the highest concentration, except at RM 0.0 at 1 m below the surface.

In August and September, total PCB concentrations in samples collected from 1 m above the bottom at RM 3.3 were higher than those in samples collected from 1 m below the surface at the same location; concentrations were similar in surface and bottom water in August samples from RM 0.0. In contrast, in November and December, total PCB concentrations in bottom water were lower than those in the surface water, except for the sample from RM 3.3 in December. Total PCB concentrations in surface water samples are compared with WQC in Section 4.2.2 (Table 4-20).

4.2.3.6 Seep water

PCB Aroclors were analyzed in unfiltered seep water samples collected at 16 locations throughout the LDW during the seep sampling event conducted in 2004 for the RI. In addition, PCB Aroclors were analyzed in seep water samples collected at three T-117 locations in 2003 and in unfiltered seep water samples collected at 17 Boeing Plant 2 locations in 1995 (Map 4-11a). Detected concentrations of PCBs in seep water samples are shown on Maps 4-11b through 4-11e.

PCBs were detected in unfiltered seep water samples collected from three of the locations sampled for the RI (SP-54 in the inlet at RM 2.2, SP-64 south of Kellogg Island, and SP-71 at RM 0.015 on the west side), from one location at T-117 (Seep 3), and from four locations at Boeing Plant 2 (SE-SWY01, SW-SWY01, SE-SWY03, and SE-84102) (Table 4-39). The maximum detected PCB concentration was 8.9 µg/L in an unfiltered sample collected at SP-54 near RM 2.2 in EAA 2. Unfiltered seep water samples may contain sediment that is suspended during sampling. PCBs were also detected in one filtered seep sample collected from SP-54 in the inlet at RM 2.2 (EAA 2). Total PCB concentrations in seep water samples are compared with WQC in Section 4.2.2 (Table 4-21).

Table 4-39. Summary of detected concentrations of PCBs in seep water

Sampling Event	Chemical	Detection Frequency		Concentration (µg/L)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs
RI – LDW-wide	**Total (not filtered or centrifuged)**					
	Aroclor 1248	2/14	14	0.092	4.7	0.017 – 0.02
	Aroclor 1254	3/14	21	0.020 J	2.3 J	0.017 – 0.02
	Aroclor 1260	2/14	14	0.16	1.9 J	0.017 – 0.02
	Total PCBs	3/14	21	0.020 J	8.9 J	nc
	Filtered[a]					
	Aroclor 1248	1/16	6	0.21	0.21	0.017
	Aroclor 1260	1/16	6	0.047	0.047	0.017
	Total PCBs	1/16	6	0.26	0.26	nc
T-117[b]	**Total (not filtered or centrifuged)**					
	Aroclor 1260	1/3	33	0.94 J	0.94 J	1.0
	Total PCBs	1/3	33	0.94 J	0.94 J	nc
Boeing Plant 2	**Total (not filtered or centrifuged)**					
	Aroclor 1254	1/17	6	0.93 J	0.93 J	1 – 1.5
	Aroclor 1260	3/17	18	1.7 J	4.6	1
	Total PCBs	4/17	24	0.93 J	4.6	nc

[a] A 1-µm filter was used during the RI sampling event.

[b] Seep 3 was resampled because it was suspected that the detected PCB concentration of 0.94 J µg/L may have been associated with suspended solids in the water sample. The sample was then centrifuged and analyzed, and PCBs were not detected at an RL of 0.033 µg/L.

LDW – Lower Duwamish Waterway
J – estimated concentration
nc – not calculated
PCB – polychlorinated biphenyl
RL – reporting limit
T-117 – Terminal 117

4.2.3.7 Summary of PCB data

A large sediment dataset is available to describe the nature and extent of PCB concentrations in the LDW, including a total of 1,327 surface sediment samples and 821 subsurface sediment samples from which lateral and vertical distributions of PCBs in LDW sediments can be described. Total PCB concentrations in surface sediment ranged from 1.6 to 220,000 µg/kg dw; the area-based 95th and 50th percentile concentrations calculated from the RI baseline dataset were 810 and 115 µg/kg dw, respectively. The highest total PCB concentrations in surface sediment were detected between RM 0.4 and RM 0.6 on the east side of the LDW (Duwamish/Diagonal EAA), in the Slip 4 EAA, between RM 3.3 and RM 3.7 on the east side (the Boeing Plant 2/ Jorgensen Forge EAA), between RM 3.5 and RM 3.7 on the west side (in the T-117 EAA), and between RM 4.8 and RM 4.9 (including the Norfolk EAA). Many samples in these areas had concentrations greater than the area-based 95th percentile of 810 µg/kg dw. The SWAC for the area between RM 0.0 and RM 5.25 was 350 µg/kg dw.

In general, the highest total PCB concentrations in subsurface samples were detected in the same areas with higher concentrations in surface sediment. At many locations in the LDW, sediment at deeper intervals had higher PCB concentrations than did sediment at uppermost intervals. There were several exceptions to this pattern, in which relatively high total PCB concentrations were detected in the uppermost interval: 1) in the navigation channel near RM 0.9 and RM 1.0, 2) near RM 2.2 on the western shoreline, 3) towards the head of Slip 4, 4) near RM 3.6 east of the navigation channel, and 5) between RM 3.5 and RM 3.7 west of the navigation channel. Discussions of how these patterns compare to the physical CSM are presented in Section 4.3.

The fish and invertebrate tissue dataset provides good spatial coverage of the waterway, represents various exposure regimes of biota, and provides representation of species of interest to the ecological risk assessment (as either receptors or prey for receptors) and to the human health risk assessment (as species and size classes used by people). In whole-body composite samples of fish, mean total PCB concentrations ranged from 71 µg/kg ww in juvenile chinook salmon in Area T1 to 4,320 µg/kg ww in shiner surfperch in Area T2. In crabs, sampling area mean total PCB concentrations in edible-meat samples (non-detected to 270 µg/kg ww) were much lower than those in hepatopancreas samples (260 to 5,500 µg/kg ww). Mean total PCB concentrations in amphipod and benthic invertebrate tissue samples were 230 and 270 µg/kg ww, respectively, and mean total PCB concentrations in softshell clam tissue samples ranged from 98 to 140 µg/kg ww. Thus, total PCB concentrations in invertebrate (except crab hepatopancreas) samples tended to be lower than in fish, and concentrations in juvenile chinook salmon were lower than concentrations in English sole and shiner surfperch. Mean lipid-normalized total PCB concentrations in fish and crab tissue ranged from non-detected to 100 mg/kg lipid, with no clear differences in concentrations among fish or crab species or tissue types (i.e., whole body, fillet, hepatopancreas, or edible meat). Mean lipid-normalized total PCB concentrations in benthic invertebrates, clams, and mussels were generally lower than in fish and crab tissue, ranging from 3.3 to 41 mg/kg lipid.

In general, the PCB congeners patterns for each tissue type were similar, except for two clam samples from Slip 4 with higher contributions of less chlorinated PCB congeners. Of the three fish species analyzed for PCB congeners, shiner surfperch had the greatest variability in congener patterns among the individual samples. The mean pattern for whole-body fish samples in 2007 showed a small increase in the contribution from more highly chlorinated PCB congeners relative to the pattern for whole-body fish samples from 2004. The TEQ potencies were similar in the English sole and shiner surfperch tissues collected in 2004, 2005, and 2007. The Aroclor composition in samples collected from 2005 to 2007 had a higher proportion of the more highly chlorinated PCB congeners than did tissues collected in 2004.

The Aroclor tissue data as a whole suggest total PCB concentrations in 2004 were higher than those in more recent years; total PCB concentrations appear to be decreasing based on both wet-weight and lipid-normalized Aroclor data. Trends based on the limited PCB congener data are more uncertain. The PCB congener data suggest that wet-weight total PCB concentrations in English sole collected from Area T1 and possibly from Area T2 may be decreasing, although the sample size is small. Total wet-weight PCB concentrations (congener sums) in crabs collected from Areas T1 and T3 were also consistently lower in 2007 than 2004. The wet-weight PCB congener data suggest that temporal trends in total PCB concentrations in shiner surfperch are uncertain. The lipid-normalized PCB congener data suggest potential downward trends in Area T3 for English sole, in Area T2 for shiner surfperch, and in Areas T1 and T3 for Dungeness crab.

Differences in total PCB concentrations in English sole and shiner surfperch tissue among the four subareas were also evaluated for both wet-weight and lipid-normalized data. For wet-weight total PCB data, area means were not significantly different in 2005 and 2007 for English sole tissue. In 2004, significant differences were identified only between the area with the lowest mean (Area T4) and the areas with the highest or second highest mean (Area T1 or T2, respectively). For shiner surfperch, total PCB concentrations did not differ among areas in 2005 but were lower in Areas T1 and T4 than in Areas T2 and T3 in 2004 and 2007. Total PCB concentrations in Pacific staghorn sculpin were significantly higher in Area T3 than in the other areas in 2004, which was the only year with sufficient data for statistical analysis. For lipid-normalized total PCB data, area means were not significantly different in 2005 and 2007 for English sole. In 2004, significant differences were identified only between the area with the lowest mean (Area T4) and the area with the highest mean (Area T1). For shiner surfperch, lipid-normalized total PCB concentrations did not differ among areas in 2004, 2005, or 2007. Results using lipid-normalized data for Pacific staghorn sculpin were the same as those based on wet-weight data; total PCB concentrations were significantly higher in Area T3 than in the other areas in 2004.

The available dataset for water samples is small relative to the tissue and sediment datasets. Surface water grab samples were collected from the LDW on four sampling dates between August and December 2005 from two locations (near Harbor Island at RM 0.0 and near the South Park Bridge at RM 3.3) at two different depths. Total PCB concentrations ranged from 0.000132 to 0.003211 µg/L with an overall mean concentration of 0.001277 µg/L. Concentrations were lowest at all locations in November 2005, when flow rates were the highest.

PCBs were detected in unfiltered seep water samples collected from three of the 16 locations sampled as part of the RI, one of the T-117 locations, and four of the Boeing Plant 2 locations. Because these samples were not filtered, these detections may represent entrained sediment at locations with elevated total PCB concentrations in sediment. Total PCBs were detected in one filtered sample collected from the inlet at RM 2.2 (EAA 2). The maximum detected PCB concentration was 8.9 µg/L in an unfiltered sample collected at a seep in EAA 2. PCBs were not analyzed in any porewater samples collected from the LDW.

4.2.4 Arsenic

This section summarizes the nature and extent of arsenic concentrations in surface sediment, subsurface sediment, tissue, surface water, seep water, and porewater. The extent of arsenic in site media is described in detail in this section because arsenic is a risk driver chemical.

4.2.4.1 Surface sediment

Surface sediment samples collected from 852 of the 1,365 locations in the RI baseline dataset were analyzed for total arsenic. Arsenic was detected in 93% of surface sediment samples analyzed for arsenic, with concentrations ranging from 1.2 mg/kg dw to 1,100 mg/kg dw and a mean concentration of 17 mg/kg dw (Table 4-40).

Table 4-40. Summary of surface sediment data for total arsenic

	Detection Frequency		Total Arsenic Concentration (mg/kg dw)					
Chemical	Ratio	%	Minimum Detection	Maximum Detection	Calculated Mean[a]	RL or Range of RLs[b]	SQS	CSL
Arsenic	794/852	93	1.2	1,100	17	3.1 – 31	57	93

[a] Calculated mean concentration is the mean of detected concentrations and one-half the RL for non-detected results.

[b] RLs based only on non-detect samples.

CSL – cleanup screening level
dw – dry weight
RL – reporting limit
SQS – sediment quality standards

Arsenic concentrations in surface sediment were much less variable than total PCB concentrations; the numerical 95th percentile of the surface sediment dataset was 30 mg/kg dw (Table 4-41; Map 4-29). Percentiles calculated on an area-basis using IDW interpolation were similar (Table 4-41; Map 4-30).

Table 4-41. Percentiles of arsenic concentrations in LDW surface sediment collected from RM 0.0 to RM 6.0

	Arsenic Concentration (µg/kg dw)[a]				
Method	Entire Dataset	25th Percentile	50th Percentile	75th Percentile	95th Percentile
Area-based	15 (SWAC)	8.6	11	14	25
Numerical	17 (mean)	7.9	11	15	30

[a] The UCL for arsenic in surface sediment was 21 mg/kg dw using the RI baseline dataset, as calculated in Appendix C (Table C.3-8). This UCL is presented as the EPC for the netfishing scenario in the HHRA (Appendix B), which uses the entire surface sediment dataset.

dw – dry weight
EPC – exposure point concentration
HHRA – human health risk assessment
LDW – Lower Duwamish Waterway
RI – remedial investigation
RM – river mile
SWAC – spatially weighted average concentration
UCL – upper confidence limit on the mean

The maximum arsenic concentration in surface sediment was detected at sampling location LDW-SS114, on the east side of the LDW between RM 3.7 and RM 3.8 (Map 4-29). Other areas with arsenic concentrations in surface sediment greater than the 95th percentile (30 mg/kg dw) were: 1) near RM 0.1 on the east side of the navigation channel, 2) between RM 0.4 and RM 0.6 in the Duwamish/Diagonal EAA, 3) at the head of Slip 1, 4) between RM 1.3 and RM 1.5 on the west side of the navigation channel, 5) at the mouth and head of Slip 3, 6) between RM 3.5 and RM 3.8 on the east side of the navigation channel in and just upstream of the Boeing Plant 2/Jorgensen Forge EAA, and 7) near RM 5.0 (Map 4-30).

Arsenic concentrations exceeded the SQS or CSL at 14 surface sediment locations in five areas of the LDW: RM 0.1 on the east side, Slip 1, RM 1.3 to RM 1.5 on the west side, Slip 3, and RM 3.7 to RM 3.8 on the east side (Map 4-29). Approximately 98% of the surface sediment samples analyzed for arsenic had concentrations less than the SQS (Figure 4-21).

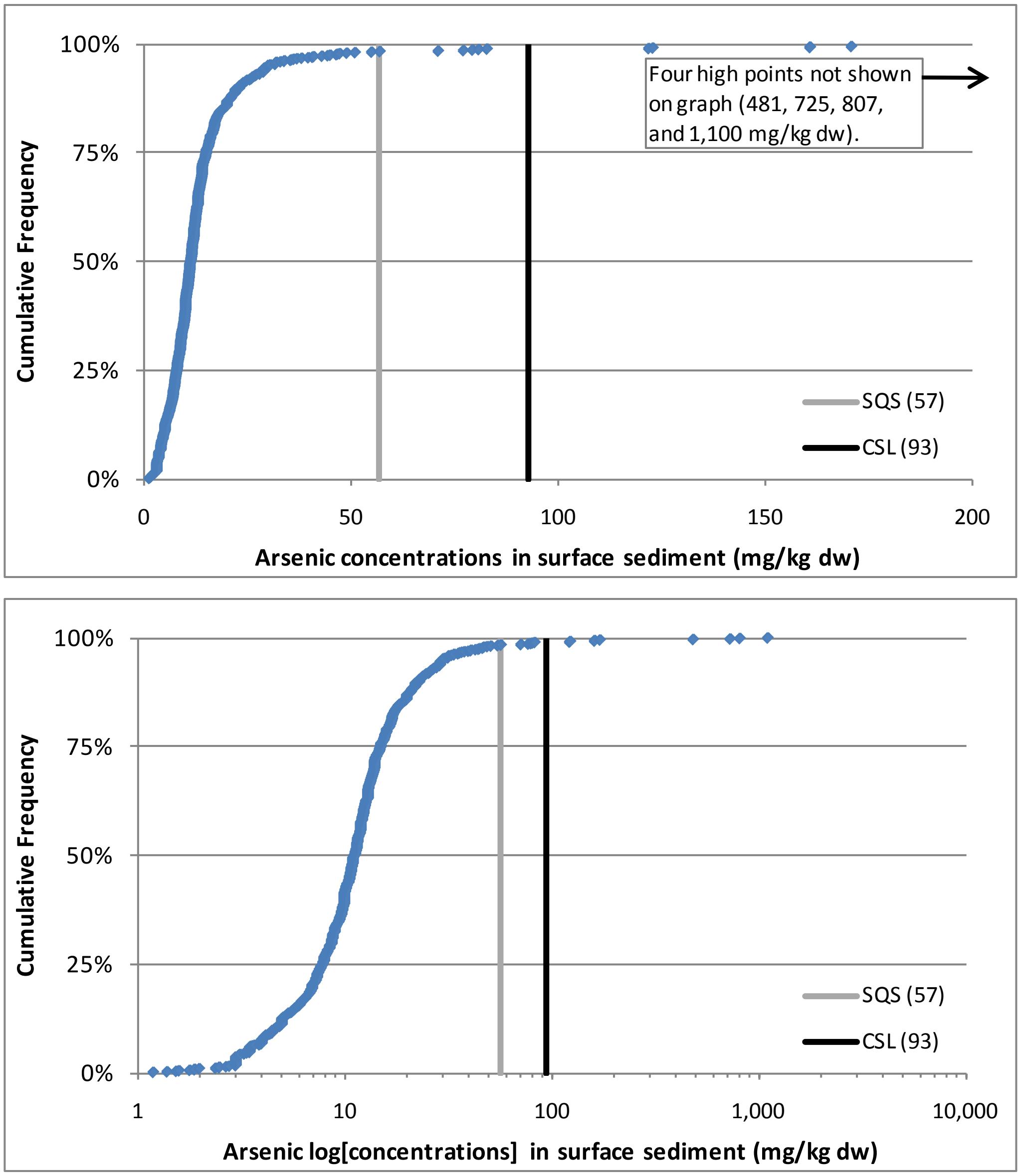

Figure 4-21. Cumulative frequency of arsenic concentrations in surface sediment (arithmetic and log-scale)

Forty-seven of the locations where surface sediment samples were collected and analyzed for arsenic were within 10 ft of where samples had previously been collected (Table 4-42). The older data were excluded from the RI baseline dataset, as described in Section 4.1.2.1 and Appendix E; however, older data for stations that were re-occupied in the RI were retained for the evaluation of changes in concentrations between older samples and newer samples only.

Table 4-42. Difference in arsenic concentrations at resampled locations, sorted by concentration difference

Approximate River Mile	Sampling Location ID		Sampling Date		Arsenic Concentration (mg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
1.3	SS-2	LDW-SS48	8/17/1993	1/18/2005	1,130	807	-323
1.4	SS-4	LDW-SS55	8/17/1993	1/24/2005	140	17.2	-123
1.5	DR123	LDW-SS57	9/14/1998	1/24/2005	52.4	35.4	-17.0
4.2	R42	LDW-SS129	10/13/1997	1/20/2005	21.1	10.6	-10.5
3.7	SD-323-S	LDW-SS110	8/17/2004	1/25/2005	32	24.7	**-7.3**
0	K-11	LDW-SS1	9/30/1991	1/17/2005	12.6	6.2	-6.4
4.3	DR286	B10b	8/26/1998	8/19/2004	10.7	5.05 J	-5.65
1.0	DR019	LDW-SS32	8/17/1998	1/18/2005	21.1	15.7	**-5.4**
4.2	R40	LDW-SS127	10/13/1997	1/20/2005	18.4	13.2	-5.2
1.7	DR097	LDW-SS63	8/20/1998	1/21/2005	14.6	10.2	-4.4
0	DR076	LDW-SS5	8/24/1998	1/17/2005	10.6	6.5	-4.1
0.2	DR035	LDW-SS12	8/11/1998	1/17/2005	16.7	13.0	**-3.7**
3.8	DR187	LDW-SS115	8/27/1998	1/25/2005	48.1	44.4	**-3.7**
0.3	DR079	LDW-SS15	8/24/1998	1/17/2005	15.1	11.5	**-3.6**
1.0	DR087	LDW-SS37	8/12/1998	1/18/2005	16.8 J	13.6	**-3.2**
3.7	R21	LDW-SS113b	10/9/1997	1/20/2005	10.8	8.3	**-2.5**
3.9	R30	LDW-SS119	10/11/1997	1/19/2005	12.4	10.9	**-1.5**
0.9	DR021	LDW-SS319	8/17/1998	10/4/2006	15.2	14.8	**-0.4**
4.1	DR238	LDW-SS125	8/27/1998	1/20/2005	8.9	8.6	**-0.3**
0.3	DUD042	LDW-SS17	11/11/1995	1/24/2005	15.0 J	14.9	**-0.1**
3.1	DR198	LDW-SS102	8/20/1998	1/24/2005	6.7	6.6	**-0.1**
0.9	DR085	LDW-SSB2b	8/31/1998	3/11/2005	16.5	16.5	**0.0**
1.4	DR028	B4b	8/17/1998	8/28/2004	9.9	10.3 J	**0.4**
2.0	R7	LDW-SS75	10/15/1997	1/21/2005	7.9	8.3	**0.4**
4.1	A11-05	LDW-SS126	8/18/1994	1/20/2005	6.5	7.3	**0.8**
4.1	06-intsed-2	SH-04	7/1/1996	8/24/2004	8.0	8.8	**0.8**
4.2	R45	LDW-SS130	10/16/1997	1/20/2005	13.9	15.0	**1.1**
1.2	DR088	LDW-SS40	8/31/1998	1/18/2005	15.4	16.7	**1.3**
2.1	DR106	LDW-SS76	8/19/1998	1/20/2005	12.7	14.5	**1.8**
4.2	DR242	SB-1	8/24/1998	8/25/2004	20	22	**2**
0.2	K-05	LDW-SS10	9/27/1991	1/17/2005	10	12.4	**2**
1.4	DR030	LDW-SS50	8/17/1998	1/24/2005	13.6	16.3	**2.7**
3.2	DR202	LDW-SS104	8/27/1998	1/25/2005	8.1	11.5	3.4
4.0	07-intsed-1	SH-02	7/1/1996	8/24/2004	7.0	11	4
3.8	R24	LDW-SS117	10/10/1997	1/20/2005	10.2	14.4	4.2

APPROXIMATE RIVER MILE	SAMPLING LOCATION ID		SAMPLING DATE		ARSENIC CONCENTRATION (mg/kg dw)		
	INITIAL	RESAMPLED	INITIAL	RESAMPLED	INITIAL	RESAMPLED	DIFFERENCE
2.2	DR113	LDW-SS81	8/19/1998	3/8/2005	13.4	18.1	4.7
1.4	DR065	LDW-SS52	8/17/1998	1/25/2005	10.3	15.5	5.2
1.9	DR131	LDW-SS70	8/13/1998	1/21/2005	8.1	14.8	6.7
3.7	DR186	LDW-SS111	8/27/1998	1/19/2005	24.9	31.7	6.8
1.4	DR160	LDW-SS51	8/12/1998	1/18/2005	9.6	16.9	7.3
3.7	SD-DUW92	SD-320-S	4/2/1996	8/16/2004	12	20	8
4.7	DR271	LDW-SS148	9/15/1998	3/9/2005	6.4	15.6	9.2
0.1	K-07	LDW-SS4	9/30/1991	1/17/2005	11.6	21.2	9.6
1.3	DR053	LDW-SS44	8/31/1998	1/21/2005	35.4 J	46.8	11.4
2.8	DR175	LDW-SS94	8/20/1998	1/21/2005	12.2	26.5	14.3
1.0	DR020	LDW-SS31	8/17/1998	1/21/2005	99.3	122	**22.7**
1.4	SS-3	LDW-SS49	8/17/1993	1/26/2005	66.0	171	105

Note: Concentrations in bold were within the range of analytical variability (≤25% increase or decrease compared with the initial concentration).

dw – dry weight

ID – identification

J – estimated concentration

At the 47 resampled surface sediment locations, arsenic concentrations decreased over time at 14 locations (30%), increased at 9 locations (19%), and were indistinguishable from the initial concentration at 24 locations (51%)[78] (Table 4-42 and Map 4-31). The area with both the highest negative (-323 and -122.8 mg/kg dw) and highest positive (+105 mg/kg dw) changes in arsenic concentrations was located between RM 1.3 and RM 1.4 on the west side of the LDW, perhaps because of high spatial variability in this area. At all other locations, changes in arsenic concentrations were relatively small (between -16.6 and +14.5 mg/kg dw).[79]

To determine if arsenic concentrations were significantly different at locations that were resampled (i.e., if the mean difference between the paired concentrations was greater than zero), a non-parametric, paired t-test (Wilcoxon signed-rank test) was used. The test was run separately for each reach of the LDW (as defined in the STM) and for the entire LDW. Data were not included for locations that were sampled less than 6 years apart.

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[78] Concentration differences at these locations were within the range of analytical variability (≤ 25% increase or decrease compared to the initial concentration).

[79] Information on heterogeneity of arsenic in sediment related to sample homogenization and laboratory variability is provided by field duplicate samples (i.e., sample splits from a grab sample after homogenization). The relative percent difference in arsenic concentrations between paired field duplicate samples in the RI (n = 11) ranged from 0 to 26%, with an average of 8%.

When the data were analyzed by reach of the LDW, the differences between initial and resampled arsenic concentrations were not significant at the 95% confidence level in any of the reaches or for the entire LDW (Table 4-43; Figure 4-22).

Table 4-43. Results of statistical analyses of differences in arsenic concentrations in surface sediment at resampled locations

REACH OF THE LDW	n[a]	MEAN CONCENTRATION (mg/kg dw)		SIGNIFICANCE (p value)
		INITIAL	RESAMPLED	
RM 0.0 to RM 2.2	27	67	55	0.86
RM 2.2 to RM 4.0	10	15	19	0.073
RM 4.0 to RM 4.8	9	13	12	0.80
Entire LDW	46	45	39	0.48

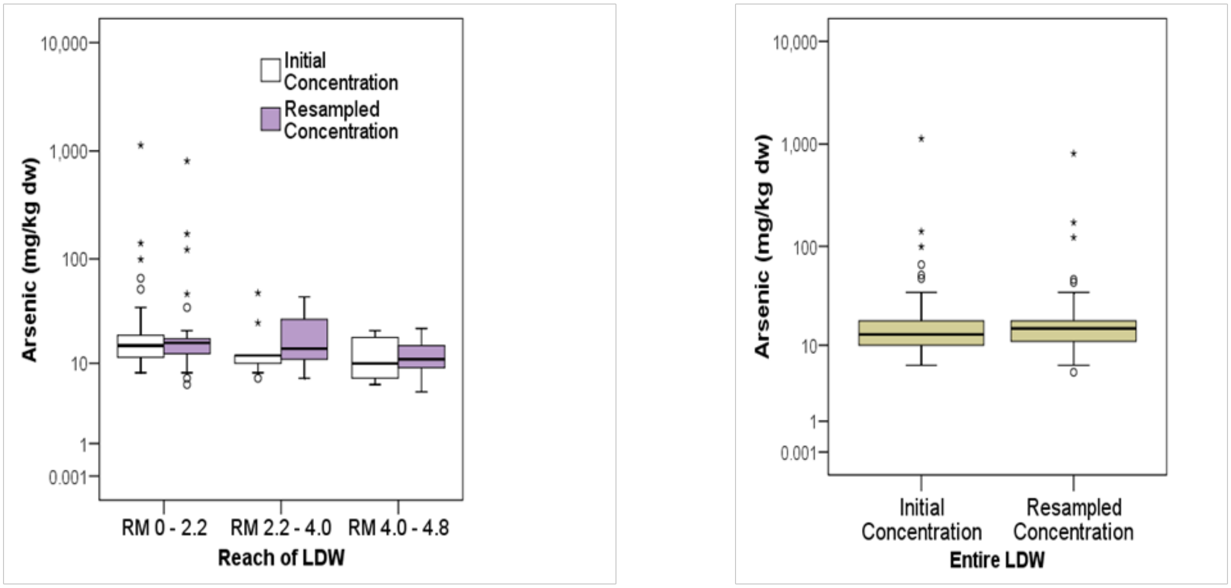
[a] One location (SD-323-S) was not included because it was resampled less than 1 year after the initial sample was collected.

dw – dry weight

LDW – Lower Duwamish Waterway

n – number of samples

RM – river mile



Note: The boxes indicate the 25th and 75th percentiles within which 50% of the data lie. The length of this box indicates the interquartile range (IQR). The line in the center of the box is the 50th percentile (median). The whiskers of the box plot indicate the maximum and minimum values from the dataset that are within 1.5 IQRs of the 25th and 75th percentile values. Outliers are indicated by a circle and have a value between 1.5 and 3 of the IQR from the upper or lower edge of the box (75th or 25th percentile). Extreme values are indicated by an asterisk and have a value > 3 times the IQR from the 75th or 25th percentile. One location (SD-323-S) was not included because it was resampled less than 1 year after the initial sample was collected.

Figure 4-22. Differences in arsenic concentrations in surface sediment at resampled locations

4.2.4.2 Subsurface sediment

Subsurface sediment cores were sampled at various intervals during the different sampling events. To calculate summary statistics, data were categorized according to specified sampling intervals, as described for total PCBs in Section 4.2.3.2.

Arsenic was detected in 82% of the subsurface sediment samples that were analyzed for arsenic, at concentrations ranging from 3.9 to 2,000 mg/kg dw (Table 4-44). Most of the subsurface arsenic data were collected in the top 4 ft of sediment cores. Arsenic data in 1- and 2-ft subsurface sampling intervals for the top 6 ft of sediment are shown graphically on Maps 4-32a through 4-32c. Data on these maps and in Table 4-44 are discussed in detail for individual cores in the following section.

Table 4-44. Summary of subsurface sediment data for total arsenic

Sampling Interval (ft)	Detection Frequency		Total Arsenic Concentration (mg/kg dw)[a]		
	Ratio	%	Minimum Detection	Maximum Detection	RL or Range of RLs[b]
All data					
Any interval	267/325	82	3.9	2,000	3.0 – 10.0
1-ft intervals[c]					
0 to 1	60/63	95	4.7	707	6.0 – 6.2
1 to 2	46/59	78	5.2	281	4.5 – 8.00
2 to 3	7/10	70	5.9	161	6.0 – 6.1
3 to 4	4/9	44	9	21	6.0 – 7
8 to 9	1/1	100	8	8	na
2-ft intervals[c]					
0 to 2	71/76	93	5.3	494	5.0 – 7
2 to 4	60/73	82	5.0	2,000	6 – 8.00
4 to 6	5/6	83	14	270	7.00
6 to 8	4/4	100	20	1,890	na
8 to 10	2/3	67	14	21	7.00

Note: Data included in the subsurface dataset are described in Section 4.1.2.2; subsurface sediment samples collected from areas that were subsequently dredged are not summarized in this table because they do not represent existing conditions.

[a] Mean concentrations were not calculated because sampling was generally focused in contaminated areas, resulting in a non-random sampling pattern, which is not likely representative of the entire LDW.

[b] RLs are presented only if there were no detected concentrations.

[c] For the calculation of summary statistics, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to the 1- or 2-ft sampling category that best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were averaged to obtain a calculated concentration for the corresponding 2-ft interval. Some of the subsurface data were collected from intervals other than 1 or 2 ft; therefore, the detection frequency ratio for "any interval" may reflect data for intervals not included in these categories.

dw – dry weight

na – not applicable

RL – reporting limit

4.2.4.3 Surface and subsurface sediment patterns

This section evaluates arsenic concentrations in surface and subsurface sediment to determine whether peak concentrations are found at the surface or at depth. The specific depths of peak concentrations within sediment cores were also evaluated. The percentiles for arsenic concentrations in sediment discussed in this section were calculated on an area-basis, using the method described in the introduction to Section 4.2 and IDW interpolation data. As discussed in Section 4.2.3.3 for total PCBs, although there is some uncertainty in the interpretation of patterns and spatial relationships between arsenic concentrations in surface and subsurface data, this analysis provides one line of evidence to understand the conceptual site model for arsenic in the LDW. Arsenic concentrations in all co-located surface and subsurface sediment samples are presented on Maps 4-33a through 4-33e. Maps 4-34 a and 4-34b show only the subsurface sediment locations with co-located surface sediment data.

Specific surface sediment locations were noted in the discussion if: 1) arsenic concentrations were higher in surface sediment samples than in sampling intervals of co-located subsurface cores, 2) the arsenic concentration in surface sediment was greater than the 75th percentile, and 3) the arsenic concentration in surface sediment was at least 25% greater than the highest arsenic concentration in any subsurface interval.

RM 0.0 to RM 1.0

Between RM 0.0 and RM 1.0, areas with the highest arsenic concentrations in surface sediment were located in Slip 1 (up to 725 mg/kg dw) and near RM 0.1 on the east side (up to 123 mg/kg dw) (Map 4-33a). These two areas also had the highest concentrations in subsurface cores collected in this section of the LDW. In the Duwamish/Diagonal EAA, between RM 0.4 and RM 0.7, arsenic concentrations in surface sediment were also higher than the area-based 95th percentile of 24 mg/kg dw prior to dredging (Map 4-33a); the maximum concentration in this area was 55 mg/kg dw.

Two subsurface cores were collected along the eastern shoreline of the LDW near RM 0.1 and RM 0.2. Peak arsenic concentrations in both cores were at depth. At LDW-SC2, the general area with the high arsenic concentrations in surface sediment, the maximum concentration (270 mg/kg dw) was in the 4-to-6-ft interval although the arsenic concentration was also relatively high (190 mg/kg dw) in the 0-to-2-ft interval. At LDW-SC4, the peak concentration (63 mg/kg dw) was detected in the 1-to-2-ft interval (Map 4-33a).

Maximum arsenic concentrations were low (non-detected to 22 mg/kg dw) in the three cores collected between RM 0.0 and RM 0.2 near the navigation channel and on the west side of the navigation channel. Peak arsenic concentrations were slightly higher (41 to 44 mg/kg dw) in two cores collected farther south along the western shoreline near RM 0.3 (LDW-SC6 and DR068). The peak concentration was at depth

(2-to-4-ft interval) in LDW-SC6 and unknown in the other core (DR068) because only one interval was sampled.

Multiple cores were collected and analyzed for arsenic between RM 0.4 to RM 0.6 in the Duwamish/Diagonal EAA (including several cores just outside of the area to the north and to the east). The highest arsenic concentration in subsurface sediment in this area was 36 mg/kg dw in DUD020. Most of the arsenic concentrations in subsurface sediment were relatively low. Four cores (LDW-SC9, DUD254, DUD020, and DUD261) had concentrations greater than the area-based 95th percentile. These peak concentrations were located at depth in these four cores. However, uncertainty is high for most of the cores because of the large 3-ft sampling intervals.

Two cores were collected within the navigation channel just outside of the Duwamish/Diagonal EAA. Peak arsenic concentrations in these cores (62 mg/kg dw in LDW-SC8 and 30 mg/kg dw in LDW-SC9) were at depth (4-to-6-ft interval and 1-to-3-ft interval, respectively). Arsenic concentrations in a third core in the navigation channel were similar throughout the core (22 to 24 mg/kg dw in LDW-SC14).

Relatively few cores were collected in the vicinity of Kellogg Island. In cores collected at three locations north and south of the island (LDW-SC12 and DR044 near RM 0.6 and LDW-SC19 near RM 1.0), arsenic concentrations were less than or equal to the 95th percentile and varied little with depth. In a fourth core, which was collected in the vicinity of the island along the western shoreline of the LDW between RM 0.5 and RM 0.6 (LDW-SC11), a peak concentration of 28 mg/kg dw was detected in the uppermost interval.

In a core collected at the head of Slip 1 (LDW-SC17), the peak arsenic concentration was at depth (170 mg/kg dw in the 1-to-2-ft interval). In three cores collected toward the mouth of Slip 1 and one core collected just downstream of Slip 1 (LDW-SC15, LDW-SC16, DR021, and LDW-SC13), arsenic concentrations were much lower (≤ 30 mg/kg dw) and did not vary much with depth. Three of these four cores had peak concentrations in the uppermost interval, and one core (DR021) had a peak concentration at depth (24 mg/kg dw in the 2-to-4-ft interval).

In summary, two areas had high arsenic concentrations (> 100 mg/kg dw) in surface sediment; cores in these areas had peak concentrations at depth. Arsenic concentrations in surface and subsurface sediment in the Duwamish/Diagonal EAA were also somewhat elevated (> 30 mg/kg dw) prior to dredging. In other areas, there were generally low arsenic concentration gradients within cores, with some peak concentrations at depth and some in the uppermost interval. In all of the cores with concentrations greater than 30 mg/kg dw (and with data for more than one interval), the peak concentrations were at depth.

RM 1.0 to RM 2.2

Between RM 1.0 and RM 2.2, the highest arsenic concentrations were detected at two locations: 1) between RM 1.3 and RM 1.5 on the west side of the LDW (up to 807 and

1,890 mg/kg dw in surface and subsurface sediment, respectively), and 2) at the head of Slip 3 (up to 81 and 2,000 mg/kg dw in surface and subsurface sediment, respectively) (Map 4-33b). In the other areas, surface and subsurface sediment concentrations were generally less than the 95th percentile.

West of the navigation channel, four cores were collected between RM 1.3 and RM 1.4 (LDW-SC25, DR054, LDW-SC26, and LDW-SC28). High arsenic concentrations were detected in subsurface sediment in these cores, with peak concentrations ranging from 250 to 1,890 mg/kg dw (Map 4-33b). The peak concentrations were at depth in each of these four cores; the two cores with the highest concentrations (760 and 1,890 mg/kg dw) each had peak concentrations at an interval of approximately 6 to 8 ft. Between RM 1.4 and RM 1.5, four cores were sampled at only one interval each, with concentrations ranging from 11 to 181 mg/kg dw; this area was dredged after the cores had been collected. In the only other core collected between RM 1.3 and RM 1.5 (LDW-SC29), arsenic concentrations were relatively low (≤ 14 mg/kg dw) and did not vary much with depth.

One core was collected from the head of Slip 3 (LDW-SC37). The peak concentration in this core (2,000 mg/kg dw) was detected at depth (2-to-4-ft interval). Two cores collected toward the mouth of Slip 3 (DR106 and LDW-SC36) had low arsenic concentrations (less than the 75th percentile) that varied only slightly with depth.

Other cores collected between RM 1.0 and RM 2.2 had relatively low arsenic concentrations (≤ 20 mg/kg dw) that varied little with depth, with four exceptions. In these four cores, arsenic concentrations were higher (ranging from 30 to 56 mg/kg dw). Peak concentrations were in the uppermost intervals at two of these locations (30 mg/kg dw in the 0-to-1-ft interval at LDW-SC24 and 56 mg/kg dw in the 0-to-2-ft interval at LDW-SC33) and in deeper intervals at the other two locations (40 mg/kg dw in the 1-to-2-ft interval at LDW-SC32 and 34 mg/kg dw in the 2-to-4-ft interval at LDW-SC21).

In summary, the two areas with relatively high arsenic concentrations in surface and subsurface sediment had peak concentrations at depth within cores. Most of the other subsurface sediment cores had relatively low arsenic concentrations that varied only slightly with depth.

RM 2.2 to RM 3.0

Arsenic concentrations were generally low in both surface and subsurface sediment samples collected between RM 2.2 and RM 3.0 (Map 4-33c). Only one area had concentrations that exceeded the area-based 95th percentile in either surface or subsurface sediment. A core collected in this area (LDW-SC45), located just downstream from Slip 4 on the east side of the LDW, had a peak arsenic concentration of 25 mg/kg dw at depth (2-to-4-ft interval). Four other subsurface cores were collected between RM 2.2 and RM 3.0 on the east side. Arsenic was not detected in the core collected at LDW-SC43 and was detected at relatively low concentrations

(≤ 20 mg/kg dw) in the other three cores (LDW-SC41, LDW-SC42, and LDW-SC44). Arsenic concentrations in these three cores varied only slightly with depth (Map 4-33c).

Four cores were collected within Slip 4 from an area that was subsequently dredged. Each of these cores was sampled at only one relatively large sampling interval (generally 4-ft intervals). Arsenic concentrations ranged from 4 to 16 mg/kg dw in these cores.

Arsenic concentrations were all less than 10 mg/kg dw in the five cores collected in the navigation channel. Only one core (DR171) was sampled at more than one interval, and arsenic concentrations were similar in both intervals in that core.

All of the 12 cores collected on the west side of the navigation channel had arsenic concentrations ≤ 20 mg/kg dw. Three of these twelve cores were sampled at more than one interval. There were minor arsenic concentration gradients with depth in these three cores.

In summary, arsenic concentrations were ≤ 25 mg/kg dw in this section of the LDW and did not vary much with depth. In the core with the highest concentration (25 mg/kg dw), the peak concentration was in the 2-to-4-ft interval.

RM 3.0 to RM 4.0

Arsenic concentrations in surface and subsurface sediment were generally less than or equal to the 75th percentile (13 mg/kg dw) between RM 3.0 and RM 4.0, except for a few areas between RM 3.5 and RM 3.8 along the eastern shoreline (Map 4-33d). The highest surface sediment concentrations were detected in the area between RM 3.7 and RM 3.8 (up to 1,100 mg/kg dw), just south of the Boeing Plant 2/Jorgensen Forge EAA. One of the cores collected in this area (LDW-SC50a) had a peak concentration in the uppermost interval (707 mg/kg dw), with decreasing concentrations at depth. The co-located surface sediment sample at this location had a higher arsenic concentration (1,100 mg/kg dw). Another sampling location in this area (DR220) also had an arsenic concentration in the co-located surface sediment sample (15.3 mg/kg dw) that was higher than the peak concentration in the subsurface core, although the concentration in surface sediment was relatively low. The third core in this area (LDW-SC51) had a peak concentration at depth (55 mg/kg dw in the 2-to-3.8-ft interval).

In surface sediment between RM 3.4 and RM 3.7, within the Boeing Plant 2/Jorgensen Forge EAA, arsenic concentrations were generally less than the 95th percentile, with a few locations with higher concentrations, up to 57 mg/kg dw. In subsurface sediment, concentrations were generally ≤ 21 mg/kg dw and many cores were sampled at only one interval. Three cores (SD-DUW53, SD-312-C, and SD-323-C) had peak concentrations higher than 21 mg/kg dw (31 mg/kg dw in the 8-to-12-ft interval, 26.7 mg/kg dw in the 1-to-2-ft interval, and 22.8 mg/kg dw in the 1-to-2-ft interval, respectively). Five of the locations with co-located surface and subsurface sediment samples had arsenic concentrations in the surface sediment that were higher than the peak concentrations in the associated subsurface sediment cores. One of these locations (SD-314-C) had a concentration in surface sediment (23 mg/kg dw) that was greater than the 75th percentile and at least 25% greater than any concentration in the subsurface intervals in the associated subsurface sediment core.

On the west side of the navigation channel, all subsurface core samples had low arsenic concentrations (≤ 11 mg/kg dw), with the exception of one sample (SB-5) collected near RM 3.3. The core at this location had a peak concentration of 22 mg/kg dw at an interval of 5 to 7.5 ft, compared with somewhat lower concentrations (8 and 8.5 mg/kg dw) in shallower intervals.

Arsenic concentrations in the navigation channel were all less than the 75th percentile, and most of the cores were sampled at only one interval. In the two cores sampled at more than one interval, concentrations varied only slightly with depth.

In summary, this section of the LDW generally had low arsenic concentrations in surface and subsurface sediment, and concentrations did not vary much with depth. The highest subsurface concentrations were in three cores on the east side of the waterway; two of these occurred in areas where surface sediments were also greater than the area-based 95th percentile concentration, and two further south were at concentrations between the area-based 75th and 95th percentile concentrations. In the core with highest arsenic concentration (707 mg/kg dw) near RM 3.8, the peak concentration was in the uppermost interval collected from 0 to 1 ft.

RM 4.0 to RM 5.0

Arsenic concentrations were all ≤ 20 mg/kg dw in both surface and subsurface sediment between RM 4.0 and RM 5.0, with the exception of concentrations in surface sediment at two locations, one in Slip 6 LDW-SS128; 26.2 mg/kg dw) and the other near RM 5.0 (NFK006; 51 mg/kg dw) (Map 4-33e).

Surface and subsurface sediment concentrations in Slip 6 were between the 75th and 95th percentile concentrations. Of the 11 cores collected in Slip 6, only two had more than one interval sampled. Many of the locations in Slip 6 had higher arsenic concentrations in the co-located surface sediment samples than in the associated subsurface cores, but only four of the locations (SB-1, SB-4, SB-6, and SB-7) had concentrations in surface sediment that were both greater than the 75th percentile and at least 25% greater than concentrations in subsurface sediment. Subsurface sediment concentrations in cores collected downstream from Slip 6 on the eastern shoreline were less than or equal to the 50th percentile concentration.

Near RM 5.0, all four subsurface cores had arsenic concentrations less than the 50th percentile. No core was collected near the area with a high concentration of 51 mg/kg dw in surface sediment; therefore, concentration gradients with depth could not be evaluated at this location.

At all other locations between RM 4.0 and RM 5.0 (i.e., in the navigation channel and to the west of the navigation channel), subsurface concentrations were less than the 75th percentile, and, in general, only one interval was sampled. Also, many of the intervals were greater than 4 ft deep, so concentration gradients with depth could not be evaluated.

In summary, most of the surface and subsurface sediment arsenic concentrations in this section of the LDW were less than the 95th percentile of 24 mg/kg dw. In the few cores that had more than one sampling interval, concentrations were ≤ 20 mg/kg dw and varied little with depth.

4.2.4.4 Tissue

Total arsenic was detected in all tissue samples in which it was analyzed. In whole-body fish, the mean total arsenic concentration was higher in English sole (3.274 mg/kg ww) than in other species (0.738 to 0.99 mg/kg ww) (Table 4-45 and Figure 4-23). Likewise, in fillet samples, the mean total arsenic concentration was higher in English sole (5.255 mg/kg ww with skin and 11 mg/kg ww without skin) compared with other species (0.274 to 0.899 mg/kg ww with skin), although there were only three fillet samples for species other than English sole.

Table 4-45. Summary of LDW-wide tissue data for total arsenic

Tissue Type	Detection Frequency		Total Arsenic Concentration (mg/kg ww)		
	Ratio	%	Minimum Detect	Maximum Detect	Mean
Fish, Whole Body					
English sole	21/21	100	2.230	4.330	3.274
Pacific staghorn sculpin	24/24	100	0.364	1.430	0.738
Shiner surfperch	27/27	100	0.715	1.4	0.99
Starry flounder	3/3	100	0.793	0.973	0.887
Fish, Fillet					
English sole (with skin)	7/7	100	3.965	6.890	5.255
English sole (without skin)	9/9	100	9.0	15	11
Pile perch (with skin)	1/1	100	0.563	0.563	0.563
Starry flounder (with skin)	1/1	100	0.899	0.899	0.899
Striped perch (with skin)	1/1	100	0.274	0.274	0.274
Crab, Edible Meat					
Dungeness crab	9/9	100	2.540	13	5.5
Slender crab	12/12	100	1.670	3.570	2.612
Crab, Hepatopancreas					
Dungeness crab	4/4	100	3.080	7.0	4.6
Slender crab	4/4	100	2.230	3.310	2.645
Crab, Whole Body (calc'd)[a]					
Dungeness crab	9/9	100	2.707 M	11.0 M	5.2
Slender crab	12/12	100	1.844 M	3.489 M	2.622

	Detection Frequency		Total Arsenic Concentration (mg/kg ww)		
Tissue Type	Ratio	%	Minimum Detect	Maximum Detect	Mean
Invertebrates, Whole Body					
Amphipod	4/4	100	0.96	1.5	1.2
Benthic invertebrates	20/20	100	0.573	17.40	2.34
Shellfish					
Mussels, wild (edible meat)	22/22	100	0.34	1.1 J	0.81
Mussels, transplanted (edible meat)	35/35	100	0.573	1.42	0.791
Soft-shell clam, non-depurated (whole body)	29/29	100	1.30 J	15.2	4.21
Soft-shell clam, depurated (whole body)	15/15	100	2.350	19.70	6.803

[a] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Arsenic concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data).

J – estimated concentration

LDW –Lower Duwamish Waterway

M – calculated whole-body concentration

na – not applicable

ww –wet weight

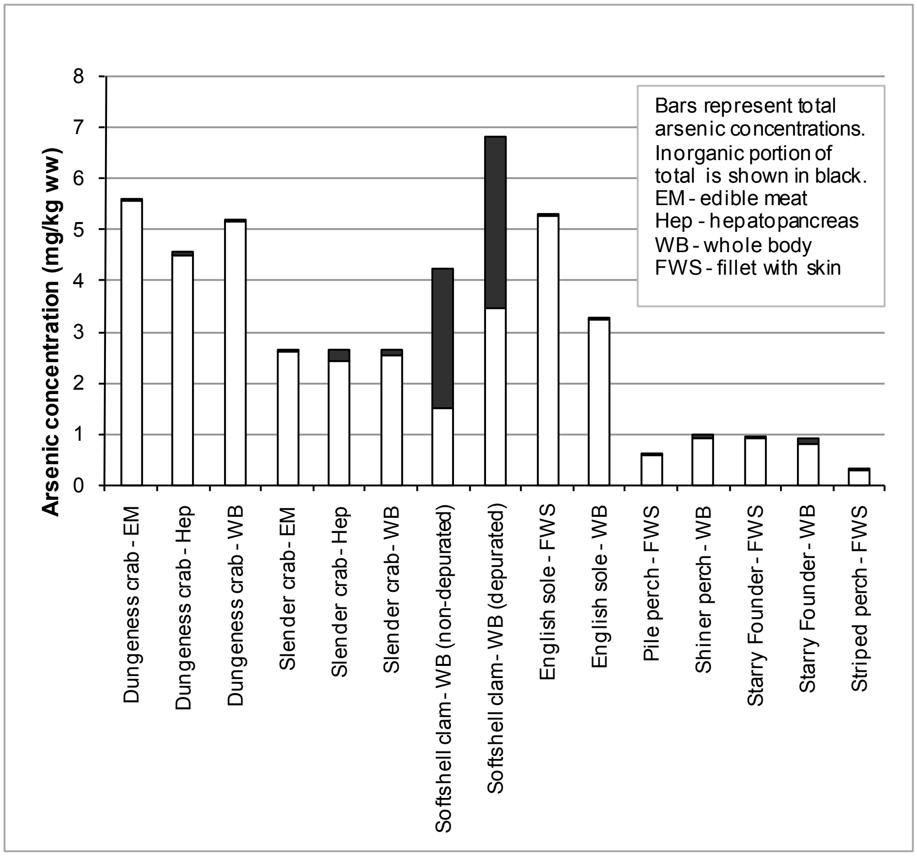


Figure 4-23. Inorganic arsenic concentrations as a percentage of total arsenic concentrations in tissue

Mean total arsenic concentrations were higher in Dungeness crabs than in slender crabs (5.2 and 2.645 mg/kg ww, respectively, in calculated whole-body samples) (Table 4-45). Mean total arsenic concentrations were lower in mussels (0.81 to 0.791 mg/kg dw) than in clams (4.21 and 6.803 mg/kg ww in non-depurated and depurated samples, respectively).[80]

The highest sample-specific total arsenic concentration in any tissue sample (19.70 mg/kg ww) was in a depurated clam sample collected from location C12 near RM 3.8 on the east side of the LDW. The non-depurated clam sample from location C12 also had a relatively high total arsenic concentration (15.2 mg/kg ww). The second highest sample-specific total arsenic concentration in any tissue sample (17.4 mg/kg ww) was in a benthic invertebrate composite sample (B3b) collected from Slip 1 (Map 4-35). The co-located sediment sample from this location had one of the highest arsenic concentrations (725 mg/kg dw) in surface sediment samples collected from the LDW. Slip 1 was also the location of a sediment core (LDW SC17) with relatively high arsenic concentrations at all intervals, to a depth of 8.5 ft.

Inorganic arsenic is the most toxic form of arsenic to humans and is a known human carcinogen, so a subset of tissue types consumed by humans (i.e., fish, crabs, and clams) was also analyzed for inorganic arsenic as part of the RI (Table 4-46). The concentration patterns of inorganic arsenic were different than those of total arsenic. The highest mean inorganic arsenic concentrations (3.37 and 2.72 mg/kg ww) were detected in depurated and non-depurated soft-shell clams, respectively. Mean inorganic arsenic concentrations in all other species were much lower, ranging from 0.002 to 0.24 mg/kg ww.

Table 4-46. Summary of LDW-wide tissue data for inorganic arsenic

	Detection Frequency		Inorganic Arsenic Concentration (mg/kg ww)			
Tissue Type	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]
Fish, Whole Body						
English sole	7/7	100	0.020	0.090	0.051	na
Shiner surfperch	8/8	100	0.020	0.160	0.070	na
Starry flounder	1/1	100	0.090	0.090	0.090	na
Fish, Fillet						
English sole (with skin)	6/7	86	0.003	0.006 J	0.004	0.003
Pile perch (with skin)	0/1	0	nd	nd	0.0050	0.010
Starry flounder (with skin)	0/1	0	nd	nd	0.002	0.003

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[80] A subset of the clam samples collected in 2007 were depurated by placing the clams in perforated plastic jars within a nylon frame, and suspending the frame in the water 3 to 5 ft beneath the boathouse (located over deep water) at Terminal 91 for 24 hours.

Tissue Type	Detection Frequency		Inorganic Arsenic Concentration (mg/kg ww)			
	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]
Striped perch (with skin)	0/1	0	nd	nd	0.0050	0.010
Crab, Edible Meat						
Dungeness crab	2/2	100	0.010	0.010	0.010	na
Slender crab	4/4	100	0.030	0.030	0.030	na
Crab, Hepatopancreas						
Dungeness crab	2/2	100	0.050	0.090	0.070	na
Slender crab	4/4	100	0.080	0.330	0.24	na
Crab, Whole Body (calc'd)[c]						
Dungeness crab	nc	nc	0.022 JM	0.035 M	0.029	na
Slender crab	nc	nc	0.046 M	0.123 M	0.098	na
Shellfish, Whole Body						
Soft-shell clam, non-depurated	23/23	100	0.132	11.3	2.72	na
Soft-shell clam, depurated	15/15	100	0.720	9.300	3.37	na

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] RLs are based only on non-detect samples.

[c] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Inorganic arsenic concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

J – estimated concentration
M – calculated whole-body concentration
na – not applicable
nc – not calculated
nd – not detected
RL – reporting limit
ww –wet weight

The amount of inorganic arsenic in fish and shellfish tissue as a percentage of total arsenic is presented in Table 4-47 and shown in Figure 4-23. Clam tissue had the highest percentage of inorganic arsenic, with a mean of 54 and 49% in the non-depurated and depurated samples analyzed for inorganic arsenic, respectively, and a range of 10 to 99% in individual samples. The ratio of inorganic to total arsenic was much lower in crab and fish tissues, typically less than 5% and always less than 15%.

Table 4-47. Inorganic arsenic as a percentage of total arsenic in tissue samples

Tissue Type	Number of Samples	Inorganic Arsenic as a Percentage of Total Arsenic		
		Mean (%)	Minimum (%)	Maximum (%)
Fish, Whole Body				
English sole	7	2	1	3
Shiner surfperch	8	7	2	13

Tissue Type	Number of Samples	Inorganic Arsenic as a Percentage of Total Arsenic		
		Mean (%)	Minimum (%)	Maximum (%)
Starry flounder	1	9	9	9
Fish, Fillet				
English sole	7	0	0	0
Pile perch	1	1	1	1
Starry flounder	1	0	0	0
Striped perch	1	2	2	2
Crab, Edible Meat				
Dungeness crab	2	0	0	0
Slender crab	4	1	1	1
Crab, Hepatopancreas				
Dungeness crab	2	2	1	3
Slender crab	4	10	2	15
Crab, Whole Body (calc'd)[a]				
Dungeness crab	2	1	1	1
Slender crab	4	4	1	5
Shellfish, Whole Body				
Soft-shell clam, non-depurated	23	54	10	99
Softshell clam, depurated	15	49	17	81

[a] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas.

For comparison, the fractions of inorganic arsenic in tissue collected from Puget Sound in other studies were well below 10%. The mean inorganic arsenic fraction for 16 fish and crab samples collected throughout Puget Sound was 0.29% (Ecology 2002a). The inorganic arsenic fractions calculated from the same study from 13 clam samples was 0.77%, much lower than the percentages found in the RI samples.

Total arsenic concentrations in tissues and sediment SWACs are presented for the four sampling areas on Map 4-35. Mean total arsenic concentrations in English sole and Pacific staghorn sculpin were similar among areas but were slightly higher in Area T1 (Map 4-35). Total arsenic concentrations in shiner surfperch tissue were similar among areas. In crab tissue, the highest total arsenic concentration was in Area T4, but only one Dungeness crab sample was collected in this area. Excluding this sample, the highest mean total arsenic concentrations in crab were in samples from Area T1.

Map 4-35 shows arsenic concentrations in benthic invertebrate and clam tissue samples at each location, along with concentrations in co-located sediment samples. The sample locations were selected to represent areas that covered the range of arsenic concentrations in sediment throughout the LDW so that a relationship between chemical concentrations in sediment and tissue could be evaluated (Windward 2004e).

Regression analyses were conducted to evaluate this relationship for arsenic concentrations in sediment and tissue (both benthic invertebrates and clams). [81]

The regression relationship for sediment and benthic invertebrates was used in the ERA (Appendix A) to estimate total arsenic concentrations in tissue throughout the LDW using the RI baseline dataset; actual tissue data were available from 20 locations. Log$_{10}$-transformed total arsenic concentrations in benthic invertebrate tissues were regressed against log$_{10}$-transformed total arsenic concentrations in sediments. Although the variance in tissue increased with sediment concentration (i.e., all of the largest residuals were found at sediment concentrations greater than 8 µg/kg dw), residuals from the regression were approximately normally distributed (R^2 of residuals against expected normal scores = 0.94), the regression was significant ($p < 0.0005$), and 78% of the variance in tissue was explained by the regression ($R^2 = 0.78$) (Figure 4-24).

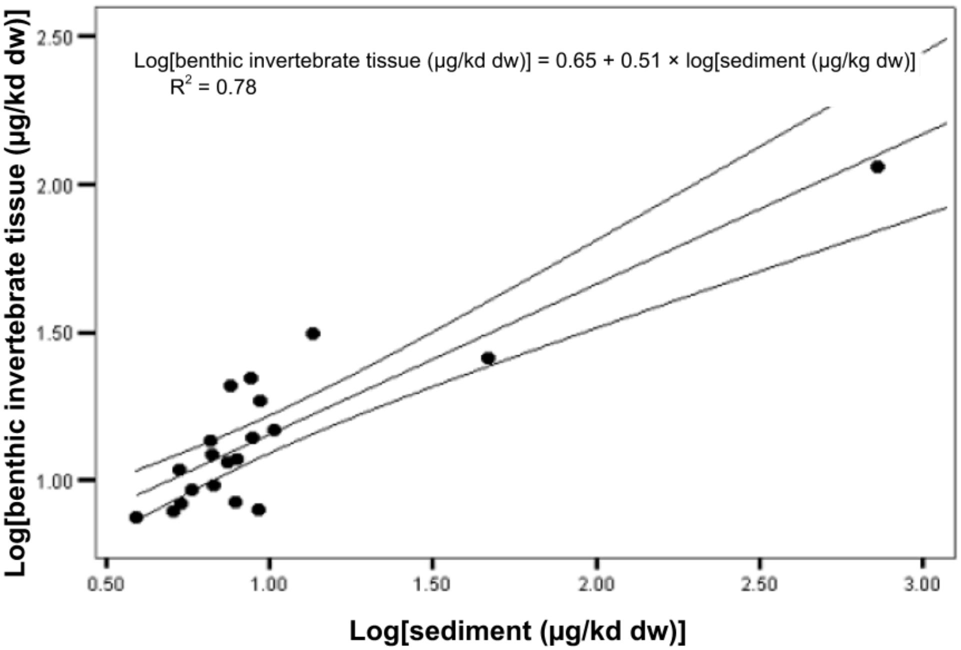

Figure 4-24. Regression equation for log$_{10}$-transformed total arsenic concentrations in co-located sediment and benthic invertebrate tissue samples, including 95% confidence intervals

The highest arsenic sediment concentration (725 mg/kg dw at location B3b in Slip 1) was co-located with the highest benthic invertebrate tissue concentration (17.4 mg/kg ww). This sediment-tissue pair could be considered an outlier in terms of its sediment

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[81] Dry weight concentrations were used for benthic invertebrate tissue to be consistent with the ERA. Wet-weight concentrations were used for clam tissue to be consistent with the HHRA; clam consumption was not a primary exposure pathway in the ERA.

concentration (Leverage value = 0.76), but it was not an outlier with respect to its tissue concentration (Studentized deleted residual = 0.58, p = 0.57). It did not have a significant effect on the estimated tissue concentration at its sediment concentration (DFFit = 0.151), the estimate of the intercept (DFbeta0 = 0.067), the estimate of the slope (DFbeta1 = 0.076), or the regression relationship in general (Cook's Distance = 0.757, p = 0.395). These results indicate that the relationship between tissue and sediment is consistent across the range of sediment concentrations that was sampled and the regression was considered a reliable means for estimating total arsenic concentrations in benthic invertebrate tissue throughout the LDW.

For clams, inorganic arsenic in tissue rather than total arsenic was more relevant for the regression because arsenic is a risk driver for human health based on the toxicity of inorganic arsenic in seafood. A logarithmic regression analysis of the relationship between inorganic arsenic concentrations in tissue and total arsenic concentrations in sediment was conducted using the 2004 and 2007 data, as described in detail in Section 8.3.2. In summary, there was a high degree of uncertainty in the relationship as represented by the spread of data and the low R^2 (0.51).

The relationships between total arsenic concentrations in fish or crab tissue and sediment were not statistically evaluated. Based on data shown on Map 4-35, the highest mean total arsenic concentrations in whole body English sole, English sole fillet, whole body Pacific staghorn sculpin, and whole body slender crab were in Area T1, which had the highest arsenic SWAC in surface sediment.

4.2.4.5 Surface water

Dissolved and total arsenic were analyzed in hundreds of water samples collected from October 1996 to June 1997 at three LDW locations as part of the King County WQA (King County 1999e). Samples at these three locations were collected at two depths: 1 m below the surface and 1 m above the sediment. Mean dissolved and total arsenic concentrations in individual samples ranged from 0.175 to 1.57 µg/L (Table 4-48). Mean dissolved and total arsenic concentrations were slightly higher at the downstream end of the LDW (at RM 1.1 and RM 1.9) than at the upstream end at RM 4.9. Arsenic concentrations in surface water samples are compared with WQC in Section 4.2.2 (Table 4-20).

Table 4-48. Summary of surface water data for arsenic

	Detection Frequency		Concentration (µg/L)		
Location ID[a]	Ratio	%	Minimum Detect	Maximum Detect	Mean
Dissolved Arsenic					
BRN (RM 1.1)	72/72	100	0.237	1.46	0.855
SWM (RM 1.9)	72/72	100	0.224	1.42	0.806
NFK (RM 4.9)	24/24	100	0.175	0.456	0.319

LOCATION ID[a]	DETECTION FREQUENCY RATIO	%	CONCENTRATION (µg/L) MINIMUM DETECT	MAXIMUM DETECT	MEAN
Total Arsenic					
BRN (RM 1.1)	167/167	100	0.292	1.53	0.931
SWM (RM 1.9)	167/167	100	0.282	1.57	0.880
NFK (RM 4.9)	56/56	100	0.183	0.868	0.493

[a] BRN, SWM, and NFK are located in the LDW near the Brandon, SW Michigan, and Norfolk CSOs, respectively.

CSO – combined sewer overflow

ID – identification

LDW – Lower Duwamish Waterway

RM – river mile

4.2.4.6 Seep water

Arsenic was analyzed in 17 seep water samples collected from the Boeing Plant 2 site in 1995, in 7 seep water samples collected from the Rhône-Poulenc site in 1995, in 1 seep water sample from the Boeing Isaacson site in 2000, in 3 seep water samples from T-117 in 2003, and in 16 seep water samples from locations throughout the LDW during the RI sampling event in 2004 (Table 4-49). The RI locations were selected based on their proximity to potential sources in consultation with EPA and Ecology. Detected concentrations of arsenic in seep water samples are shown on Maps 4-11b through 4-11e. Arsenic concentrations in seep water samples are compared with WQC in Section 4.2.2 (Table 4-21).

Table 4-49. Concentrations of arsenic in LDW-wide seep water samples

CHEMICAL	DETECTION FREQUENCY RATIO	%	CONCENTRATION (µg/L) MINIMUM DETECT	MAXIMUM DETECT	CALCULATED MEAN[a]	RL OR RANGE OF RLs[b]
Boeing Plant 2						
Dissolved arsenic	10/10	100	6	10	9	na
Total arsenic	16/17	94	2	20	9	5
Rhône-Poulenc						
Total arsenic	1/7	14	31.0	31.0	nc	nr
RI – LDW-wide						
Dissolved arsenic	16/16	100	0.054	253	22	na
Total arsenic	13/13	100	0.058	287	28	na
T-117						
Dissolved arsenic	0/3	0	nd	nd	25	50
Total arsenic	0/1	0	nd	nd	25	50
Boeing Isaacson						
Dissolved arsenic	1/1	100	7	7	7	na

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] RLs are based only on non-detect samples.

LDW – Lower Duwamish Waterway
na – not applicable
nc – not calculated
nd – not detected
nr – not reported (RLs for non-detects not reported for this sampling event)
RL – reporting limit

At Boeing Plant 2, total arsenic concentrations in seep water were generally low, ranging from 2 to 20 µg/L; dissolved arsenic concentrations ranged from 6 to 10 µg/L. At Boeing Isaacson, the single seep sample had a dissolved arsenic concentration of 7 µg/L. At Rhône-Poulenc, total arsenic was detected in one of the seven seep water samples at a concentration of 31 µg/L; RLs were not available for the samples with non-detects. At T-117, arsenic was not detected in seep water samples at an RL of 50 µg/L.

During the RI sampling event in 2004, the highest dissolved arsenic concentrations were detected in seep water collected at location SP-76 (Slip 1 at RM 1.0 E) and location SP-61 (RM 1.5 W) at concentrations of 253 µg/L and 72.4 µg/L, respectively (Map 4-36). These two sampling locations with the highest arsenic concentrations in seep water were near locations with high arsenic concentrations in surface sediment (725 mg/kg dw at B3b in Slip 1 and 161 mg/kg dw at LDW-SS56 near RM 1.4 W). At the other 14 seep locations sampled for the RI, arsenic concentrations (total and dissolved) were less than 7 µg/L.

4.2.4.7 Porewater

Arsenic was analyzed in 16 unfiltered porewater samples collected from 14 locations at the Rhône-Poulenc site in 2004 and in 15 porewater samples collected throughout the LDW during the EPA SI, as shown on Map 4-12a. Detected concentrations of arsenic in porewater samples are shown on Maps 4-12b through 4-12d. Total arsenic was detected in 12 of the 15 porewater samples collected during the EPA SI at concentrations ranging from 26 to 114 µg/L, with a mean concentration of 50 µg/L (Table 4-50). Arsenic concentrations in porewater samples are compared with WQC in Section 4.2.2 (Table 4-22).

Table 4-50. Arsenic concentrations in EPA SI porewater samples

SAMPLING LOCATION	RIVER MILE	TOTAL ARSENIC CONCENTRATION (µg/L)[a]
DR055	0.1	63
DR038	0.4	67
DR018	1.0	59
DR096	1.7	74
DR147	1.9	93

SAMPLING LOCATION	RIVER MILE	TOTAL ARSENIC CONCENTRATION (µg/L)[a]
DR133	2.0	43
DR109	2.1	103
DR140	2.4	10 U
DR181	2.9	34
DR228	3.4	31
DR260	4.1	36
DR244	4.2	114
DR262	4.2	26
DR298	5.3	10 U
DR301	5.4	30 U

EPA – US Environmental Protection Agency
SI – site inspection

4.2.4.8 Summary of arsenic data

Arsenic concentrations in surface sediment ranged from 1.2 mg/kg dw to 1,100 mg/kg dw; the 95th and 50th percentile concentrations calculated from the RI baseline dataset were 30 and 15 mg/kg dw, respectively. The highest concentration (1,100 mg/kg dw) was detected between RM 3.7 and RM 3.8 on the east side of the navigation channel. Surface sediment samples collected from several other areas, including from RM 0.4 to RM 0.7 on the east side and from RM 1.3 to RM 1.5 on the west side, had high arsenic concentrations (i.e., greater than the 95th percentile).

Areas with relatively high arsenic concentrations in surface sediment generally also had high concentrations in subsurface sediment. Specific areas with high subsurface concentrations were: 1) Slip 3 (2,000 mg/kg dw), 2) between RM 1.3 and RM 1.4 (1,890 mg/kg dw), 3) between RM 3.7 and RM 3.8 on the eastern shoreline (707 mg/kg dw), 4) Slip 1 (170 mg/kg dw), and 5) near RM 0.1 along the eastern shoreline (270 mg/kg dw). In all of these areas, peak arsenic concentrations were detected at depth within the cores, except for the core collected between RM 3.7 and RM 3.8, which had a peak concentration in the uppermost interval of the core.

In tissue, mean total arsenic concentrations were highest in depurated softshell clams (6.803 mg/kg ww), Dungeness crab tissue (5.5 mg/kg ww in edible meat and 5.2 mg/kg ww in whole-body samples), and in English sole fillet tissue (5.255 mg/kg ww with skin and 11 mg/kg ww without skin). Soft-shell clams had the highest mean inorganic arsenic concentrations (2.27 and 3.37 mg/kg ww); mean inorganic arsenic concentrations in other tissue types were ≤ 0.24 mg/kg ww.

Regression analyses showed a significant and strong positive relationship between log-transformed total arsenic concentrations in benthic invertebrate tissue and co-located samples ($R^2 = 0.78$, $p < 0.0005$). The relationship between inorganic arsenic concentrations in clam tissue and total arsenic concentrations in co-located sediment using both the 2004 and 2007 data was considered too uncertain to provide a reliable estimate of the tissue-sediment relationship (see Section 8.3.2).

In surface water, mean dissolved and total arsenic concentrations were higher at the two downstream locations (RM 1.1 and RM 1.9), with concentrations ranging from 0.806 to 0.931 µg/L, than at the upstream location at RM 4.9 (with concentrations ranging from 0.319 to 0.493 µg/L). The highest dissolved arsenic concentrations in seep water (72.4 and 253 µg/L) were detected near RM 1.5 on the west side and in Slip 1, respectively. These areas also had high arsenic concentrations in surface sediment. Total arsenic concentrations in porewater samples collected from 15 locations throughout the LDW ranged from 26 to 114 µg/L.

4.2.5 cPAHs

This section summarizes the nature and extent of cPAH concentrations in surface sediment, subsurface sediment, tissue, and surface water. cPAHs were detected in one porewater sample but were not detected in any seep water samples. Porewater results are presented for individual PAH compounds in Section 4.2.9.6. Total cPAH concentrations in sediment, tissue, and surface water (estimated from SMPD samples) were calculated based on concentrations of seven individual PAHs (benzo(a)pyrene, benzo(b)fluoranthene, benzo(a)anthracene, benzo(k)fluoranthene, indeno(1,2,3-cd)pyrene, dibenz(a,h)anthracene, and chrysene) and their carcinogenic potency relative to benzo(a)pyrene, using PEFs from California EPA (California EPA 1994) as described in Section 4.1.4 and in Appendix E, Section E3. None of the individual cPAH compounds were detected in seep water samples. cPAHs were detected in one porewater sample (data are presented in Section 4.2.9.6). Data for individual and total PAHs are summarized in Section 4.2.9.

4.2.5.1 Surface sediment

Surface sediment samples from 828 of the 1,365 locations in the RI baseline dataset were analyzed for at least one individual cPAH compound. Most of the 828 samples were analyzed for all cPAH compounds; samples from 17 locations were analyzed for only a subset of the individual cPAH compounds.[82] At least one cPAH compound was detected in 94% of sediment samples analyzed for PAHs, with cPAH concentrations ranging from 9.7 to 11,000 µg/kg dw (Table 4-51). The highest RLs (ranging from 65 to 130 µg/kg dw) were from samples collected in 1995. More sensitive analytical methods (e.g., EPA 8270-SIM) were used for recent data collection efforts, achieving RLs of 45 µg/kg dw or lower. The wide range of RLs in the surface sediment dataset for cPAHs is attributed to the use of different analytical methods (EPA 8270 full-scan or EPA 8270-SIM) and required analytical dilutions for some samples.

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[82] The cPAHs in the total cPAH calculation always included at least five cPAH compounds.

Table 4-51. Summary of surface sediment data for cPAHs

CHEMICAL	DETECTION FREQUENCY		CONCENTRATION (µg/kg dw)[a]			
	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	CALCULATED MEAN[b]	RANGE OF NON-DETECTS[c]
cPAHs	780/828	94	9.7 J	11,000	500	9.0 – 130

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

[b] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[c] Non-detect values were calculated for samples with no detected cPAH compounds. Non-detect values represent the sum of one-half the RL multiplied by the PEF for each compound.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw – dry weight
J – estimated concentration
PEF – potency equivalency factor
RL – reporting limit

The numerical 95th percentile of cPAH concentrations in the surface sediment dataset was 1,500 µg/kg dw (Table 4-52; Figure 4-25; Map 4-37). Area-based percentiles calculated using IDW interpolation were slightly lower than the numerical percentiles (Table 4-52; Map 4-38).

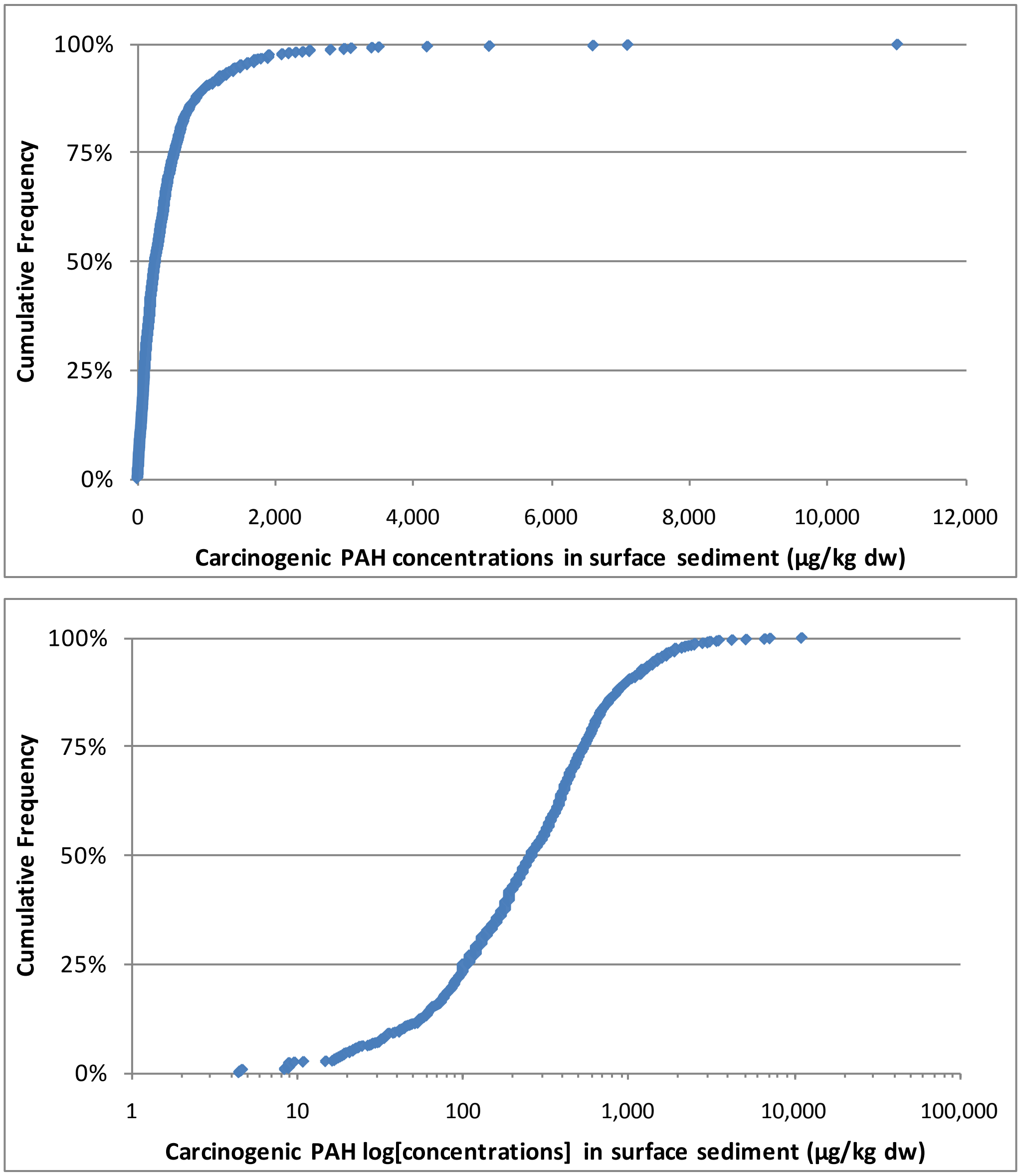
Table 4-52. Percentiles of cPAH concentrations in LDW sediment collected from RM 0.0 to RM 6.0

METHOD	CONCENTRATION (µg/kg dw)[a, b]				
	ENTIRE DATASET	25TH PERCENTILE	50TH PERCENTILE	75TH PERCENTILE	95TH PERCENTILE
Area-based	380 (SWAC)	130	260	460	1,100
Numerical	500 (mean)	100	260	540	1,500

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

[b] The UCL for cPAHs in surface sediment was 570 µg/kg dw using the RI baseline dataset as calculated in Appendix C (Table C.3-8). This UCL is presented as the EPC for the netfishing scenario in the HHRA (Appendix B), which uses the entire surface sediment dataset.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw – dry weight
EPC – exposure point concentration
HHRA – human health risk assessment
LDW – Lower Duwamish Waterway
RI – remedial investigation
RM – river mile
SWAC – spatially weighted average concentration
UCL – upper confidence limit on the mean



Note: Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

Figure 4-25. Cumulative frequency of cPAH concentrations in surface sediment (arithmetic and log-scale graphs)

The maximum cPAH concentration was detected at a location near RM 3.6 in the T-117 EAA (T117-SE-37-G). Other areas with the highest cPAH concentrations (greater than both the numerical and area-based 95th percentiles) were: 1) in the Slip 4 EAA (7,100 and 5,100 µg/kg dw at C8 and DR178, respectively), 2) in the RM 3.8 EAA on the eastern shoreline (6,600 and 3,500 µg/kg dw at R23 and R22, respectively), 3) near RM 0.6 on the western shoreline (4,200 µg/kg dw at LDW-SS312), 4) at RM 0.0 near Harbor Island (2,200 µg/kg dw at LDW-SS301), and 5) in the Duwamish/Diagonal EAA (2,900 µg/kg dw at DUD005). Other sample locations or areas with cPAH concentrations greater than the numerical or area-based 95th percentiles are shown on Maps 4-37 and 4-38, respectively.

The primary component of the cPAH concentration was benzo(a)pyrene, which accounted for 66% of the carcinogenic potency of PAHs based on an average in all surface sediment samples (standard deviation of 7%). Benzo(b)fluoranthene accounted for an average of 9% of the carcinogenic potency of PAHs (standard deviation of 3%), and the remaining five individual cPAHs each accounted for 7% or less of the carcinogenic potency of PAHs.

Forty-four locations were resampled and analyzed for cPAHs between 2004 and 2006 at locations within 10 ft of the original samples;[83] most of the original samples were collected in 1997 or 1998. Of the 44 resampled surface sediment locations, cPAH concentrations decreased during this 6-to-9-year time period at 34 locations (77%), increased at 4 locations (9%), and were indistinguishable from the initial concentration at 6 locations (14%)[84] (Table 4-53; Map 4-39). The highest increase in cPAH concentration (1,830 µg/kg dw) was located in Slip 6.[85]

Table 4-53. Difference in chemical concentrations of cPAHs at resampled locations sorted by concentration difference

	Sampling Location ID		Sampling Date		Concentration (µg/kg dw)[a]		
River Mile	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
4.2	R40	LDW-SS127	10/13/1997	1/20/2005	31,000	640	-30,360
4.2	R42	LDW-SS129	10/13/1997	1/20/2005	8,600	860	-7,740
4.2	R45	LDW-SS130	10/16/1997	1/20/2005	4,800	370	-4,430
3.8	DR187	LDW-SS115	8/27/1998	1/25/2005	5,600	2,400	-3,200
2.8	DR175	LDW-SS94	8/20/1998	1/21/2005	2,000	100	-1,900
1.0	DR019	LDW-SS32	8/17/1998	1/18/2005	2,100	340	-1,760
1.0	DR020	LDW-SS31	8/17/1998	1/21/2005	1,900	600	-1,300
1.3	DR053	LDW-SS44	8/31/1998	1/21/2005	1,700 J	670	-1,030
1.0	DR087	LDW-SS37	8/12/1998	1/18/2005	1,200	210	-990
1.7	DR097	LDW-SS63	8/20/1998	1/21/2005	1,000	190	-810
1.3	SS-2	LDW-SS48	8/17/1993	1/18/2005	2,160 J	1,400	-760

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[83] The older data within 10 ft of the newer sample were excluded from the baseline dataset, as described in Section 4.1.2.1.

[84] Concentration differences at these locations were within the range of analytical variability (≤ 25% increase or decrease compared to the initial concentration).

[85] Information on heterogeneity of cPAHs in sediment related to sample homogenization and laboratory variability is provided by field duplicate samples (i.e., sample splits from a grab sample after homogenization). The relative percent difference in cPAH concentrations between paired field duplicate samples in the RI (n = 11) ranged from 13 to 156%, with an average of 48%.

River Mile	Sampling Location ID		Sampling Date		Concentration (µg/kg dw)[a]		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
0.1	K-07	LDW-SS4	9/30/1991	1/17/2005	1,000 J	270	-730
1.4	SS-3	LDW-SS49	8/17/1993	1/26/2005	1,080	400	-680
0.2	DR035	LDW-SS12	8/11/1998	1/17/2005	840	200	-640
0.3	DUD042	LDW-SS17	11/11/1995	1/24/2005	1,080 J	440	-640
1.4	DR065	LDW-SS52	8/17/1998	1/25/2005	700	160	-540
1.5	DR123	LDW-SS57	9/14/1998	1/24/2005	770	350 J	-420
0	K-11	LDW-SS1	9/30/1991	1/17/2005	530 J	130	-400
2.1	DR106	LDW-SS76	8/19/1998	1/20/2005	510	110	-400
1.4	DR160	LDW-SS51	8/12/1998	1/18/2005	540	170	-370
1.4	SS-4	LDW-SS55	8/17/1993	1/24/2005	559	190	-369
3.7	SD-323-S	LDW-SS110	8/17/2004	1/25/2005	600	250	-340
0.2	K-05	LDW-SS10	10/14/1991	1/17/2005	800 J	480	-320
0.3	DR079	LDW-SS15	8/24/1998	1/17/2005	460	140	-320
1.4	DR028	B4b	8/17/1998	8/28/2004	600	300	-300
0.9	DR021	LDW-SS319	8/17/1998	10/4/2006	830	560 J	-270
4.7	DR271	LDW-SS148	9/15/1998	3/9/2005	430	230	-200
3.9	R30	LDW-SS119	10/11/1997	1/19/2005	420	260	-160
1.2	DR088	LDW-SS40	8/31/1998	1/18/2005	230	95	-135
0.9	DR085	LDW-SSB2b	8/31/1998	3/11/2005	390	260	-130
1.9	DR131	LDW-SS70	8/13/1998	1/21/2005	500	410	**-90**
3.1	DR198	LDW-SS102	8/20/1998	1/24/2005	150	61	-89
3.2	DR202	LDW-SS104	8/27/1998	1/25/2005	130	52 J	-78
4.3	DR286	B10b	8/26/1998	8/19/2004	100	24 J	-76
2.0	R7	LDW-SS75	10/15/1997	1/21/2005	170	130	**-40**
3.8	R24	LDW-SS117	10/10/1997	1/20/2005	100	78	**-22**
3.7	R21	LDW-SS113b	10/9/1997	1/20/2005	190	190	**0**
1.4	DR030	LDW-SS50	8/17/1998	1/24/2005	400	410	**10**
4.1	DR238	LDW-SS125	8/27/1998	1/20/2005	160	170	**10**
4.1	A11-05	LDW-SS126	8/18/1994	1/20/2005	130	180	**50**
0	DR076	LDW-SS5	8/24/1998	1/17/2005	18	89	71
2.2	DR113	LDW-SS81	8/19/1998	3/8/2005	140	270	130
3.7	DR186	LDW-SS111	8/27/1998	1/19/2005	1,200	1,900	700
4.2	DR242	SB-1	8/24/1998	8/25/2004	470 J	2,300 J	1,830

Note: Concentrations in **bold** were within the range of analytical variability (≤25% increase or decrease compared with the initial concentration). All seven cPAH compounds were analyzed in all samples presented in this table.

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

cPAH – carcinogenic polycyclic aromatic hydrocarbons
dw – dry weight
ID – identification
J estimated concentration

To determine if cPAH concentrations were significantly different at locations that were resampled (i.e., if the mean difference between the paired concentrations was greater than zero), a non-parametric, paired t-test (Wilcoxon signed-rank test) was used. The test was run separately for each reach of the LDW (as defined in the STM) and for the entire LDW. Data were not included for locations that were sampled less than six years apart.

When the data were analyzed by reach of the LDW, the differences between initial and resampled cPAH concentrations were significant at the 95% confidence level in the reach between RM 0.0 and RM 2.2 (Table 4-54; Figure 4-26). When data were analyzed for the entire LDW, differences between the initial and resampled cPAH concentrations were significant at the 95% confidence level.

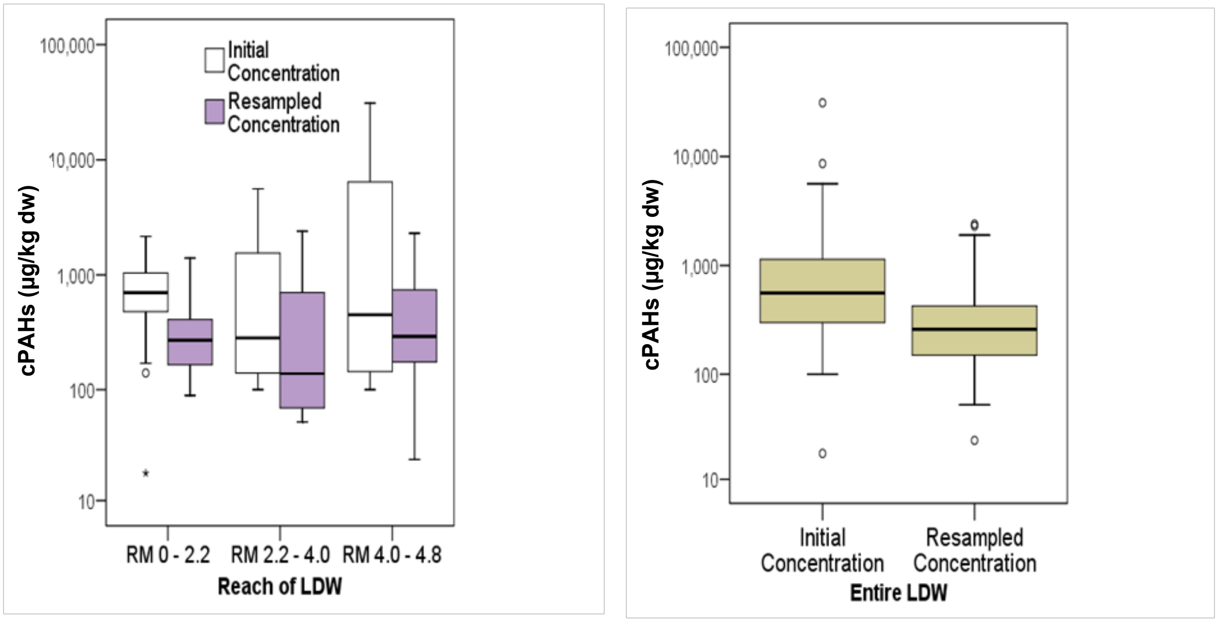
Table 4-54. Results of statistical analyses of differences in cPAH concentrations in surface sediment at resampled locations

REACH OF THE LDW	n[a]	MEAN CONCENTRATION (µg/kg dw)[b]		SIGNIFICANCE (p value)
		INITIAL	RESAMPLED	
RM 0.0 to RM 2.2	27	820	330	**< 0.005**
RM 2.2 to RM 4.0	8	1,200	630	0.13
RM 4.0 to RM 4.8	8	5,700	600	0.16
Entire LDW	43	1,800	440	**< 0.005**

[a] One location (SD-323-S) was not included because it was resampled less than 1 year after the initial sample was collected.

[b] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw – dry weight
LDW – Lower Duwamish Waterway
n – number of samples
RM – river mile
Bold identifies p values that are significant at the 95% confidence level.



Note: Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3. The boxes indicate the 25th and 75th percentiles within which 50% of the data lie. The length of this box indicates the interquartile range (IQR). The line in the center of the box is the 50th percentile (median). The whiskers of the box plot indicate the maximum and minimum values from the dataset that are within 1.5 IQRs of the 25th and 75th percentile values. Outliers are indicated by a circle and have a value between 1.5 and 3 of the IQR from the upper or lower edge of the box (75th or 25th percentile). Extreme values are indicated by an asterisk and have a value > 3 times the IQR from the 75th or 25th percentile. One location (SD-323-S) was not included because it was resampled less than 1 year after the initial sample was collected.

Figure 4-26. Differences in cPAH concentrations in surface sediment at resampled locations

4.2.5.2 Subsurface sediment

Subsurface sediment cores were sampled at various intervals during the different sampling events. To calculate summary statistics, data were categorized according to specified intervals, as described for total PCBs in Section 4.2.3.2.

cPAHs were detected in 83% of the subsurface samples in which they were analyzed, at concentrations ranging from 18 to 7,000 µg/kg dw (Table 4-55). Most of the subsurface cPAH data were collected in the top 4 ft of the sediment cores. cPAH data in the 1- and 2-ft intervals for the top 6 ft of sediment are shown on Maps 4-40a through 4-40c. Data on these maps and in Table 4-55 are discussed in detail for individual cores in the following section (Section 4.2.5.3).

Table 4-55. Summary of subsurface sediment data for cPAHs

Sampling Interval (ft)	Detection Frequency		Concentration (µg/kg dw)[a, b]		
	Ratio	%	Minimum Detect	Maximum Detect	Non-Detect or Range of Non-Detects[c]
All Data					
Any interval	252/304	83	18	7,000	3.9 – 400
1-ft Intervals[d]					
0 to 1	60/64	94	19 J	4,400	18 – 400
1 to 2	34/45	76	19 J	7,000	17 – 35
2 to 3	6/10	60	18	530	18 – 90
3 to 4	2/9	22	19 J	52 J	17 – 36
4 to 5	1/1	100	75 J	75 J	na
2-ft Intervals[d]					
0 to 2	71/74	96	41.6 J	4,900	18
2 to 4	56/73	77	18 J	5,600	17 – 35
4 to 6	9/10	90	54.1	1,300	48
6 to 8	5/5	100	48	4,000 J	na
8 to 10	4/5	80	93 J	290	4.89

Note: Data included in the subsurface dataset are described in Section 4.1.2.2; subsurface sediment samples collected from areas that were subsequently dredged are not summarized in this table because they do not represent existing conditions.

[a] Mean concentrations were not calculated because sampling was generally focused in contaminated areas, resulting in a non-random sampling pattern, which is not likely representative of the entire LDW.

[b] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

[c] Non-detect values were calculated for samples with no detected cPAH compounds. Non-detect values represent the sum of one-half the RL multiplied by the PEF for each compound.

[d] For the calculation of summary statistics in this table, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to the 1- or 2-ft sampling category that best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were averaged to obtain a calculated concentration for the corresponding 2-ft interval. Some of the subsurface data were collected from intervals other than 1 or 2 ft; therefore, the detection frequency ratio for "any interval" may reflect data for intervals not included in these categories.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw – dry weight
J – estimated concentration
na – not applicable
nc – not calculated
PEF – potency equivalency factor
RL – reporting limit

4.2.5.3 Surface and subsurface sediment patterns for cPAHs

cPAH concentrations in surface and subsurface sediment were evaluated to determine whether peak concentrations were detected in the uppermost interval of sediment cores or at depth. The specific depths of peak concentrations within sediment cores were also evaluated to assess patterns (Maps 4-41a through 4-41e). The percentiles of cPAH concentrations in sediment discussed in this section were calculated on an area basis using the method described in the introduction to Section 4.2 and the results of the IDW interpolation. As discussed in Section 4.2.3.3 for total PCBs, although there is some uncertainty in the interpretation of patterns and spatial relationships between cPAH concentrations in surface and subsurface data, this analysis provides one line of evidence to understand the conceptual site model for cPAHs in the LDW. Specific surface sediment locations were noted in the discussion if: 1) cPAH concentrations were higher in surface sediment samples than in sampling intervals of co-located subsurface cores, 2) the cPAH concentration in surface sediment was greater than the 75th percentile concentration, and 3) the cPAH concentration in surface sediment was at least 25% greater than the highest arsenic concentration in any subsurface interval. cPAH concentrations in all co-located surface and subsurface sediment samples are presented on Maps 4-41a through 4-41e. Maps 4-42a and 4-42b show only the subsurface sediment locations with co-located surface sediment data.

RM 0.0 to RM 1.0

In this reach, cPAH concentrations in surface and subsurface sediment were variable; many surface and subsurface sediment samples from RM 0.0 to RM 1.0 had cPAH concentrations higher than the area-based 75th percentile of 460 µg/kg dw. Areas with the highest cPAH concentrations (i.e., greater than the area-based 95th percentile of 1,100 µg/kg dw) in both surface and subsurface sediment included RM 0.4 to RM 0.6 (the Duwamish/Diagonal EAA), near RM 0.1 on the west side of the navigation channel, near RM 0.5 on the western shoreline, and in Slip 1 (Map 4-41a).

Six cores were collected north of RM 0.3. Three of these cores, along the eastern shoreline and near the navigation channel, had relatively low cPAH concentrations in subsurface sediment. Two of the cores, one near Harbor Island and one near RM 0.3 on the west side, had higher cPAH concentrations, with peak values detected in the uppermost interval (620 µg/kg dw at LDW-SC1 and 560 µg/kg dw at LDW-SC6). The remaining core (LDW-SC5), near RM 0.1, had a peak cPAH concentration of 1,900 µg/kg dw at depth (in the 1-to-2.2-ft interval).

In the vicinity of the Duwamish/Diagonal EAA, the highest surface and subsurface cPAH concentrations were generally located near RM 0.45. Three of the four cores with high subsurface cPAH concentrations (i.e., greater than the area-based 95th percentile) in the vicinity of the EAA were collected near RM 0.45; two of the cores in this area had peak concentrations in the uppermost interval (1,700 µg/kg dw at DR008 and 1,420 µg/kg dw at DUD256), and one core had a peak cPAH concentration at depth (1,630 µg/kg dw at DUD254). The depth of the peak in the fourth core (1,560 µg/kg dw at DUD258 near RM 0.5) is not known because only one interval (0 to 3 ft) was analyzed. The other cores collected within and close to the EAA had peak cPAH concentrations ranging from 35 to 940 µg/kg dw. Some of these peak cPAH concentrations were in the uppermost intervals, and some were at depth, with no clear spatial pattern. At two sampling locations (LDW-SC7 at RM 0.4 and DUD027 at RM 0.6), the cPAH concentrations (1,010 and 860 µg/kg dw, respectively) in co-located surface sediment samples were higher than the peak concentrations in the associated subsurface cores. There is high uncertainty associated with interpretation of data from most of the samples collected within the Duwamish/Diagonal EAA because they were collected at 3-ft intervals.

In Slip 1, the core collected at the head of the slip (LDW-SC17) had a relatively high cPAH concentration (2,400 µg/kg dw) at an interval of 6 to 8.6 ft; concentrations were also high (1,800 µg/kg dw) in the uppermost interval. In a core collected near the mouth of the slip (LDW-SC16), the peak cPAH concentration (1,300 µg/kg dw) was detected at depth (in the 4-to-6-ft interval). Peak cPAH concentrations were also at depth (in the 2-to-4-ft interval) in two other cores collected at the mouth of Slip 1 (550 µg/kg dw at LDW-SC15 and 760 µg/kg dw at DR021); concentrations in the uppermost intervals of these cores were only slightly lower. The core collected just downstream from Slip 1 (LDW-SC13) had a peak concentration of 1,200 µg/kg dw in the 2-to-4-ft interval.

At the remaining locations between RM 0.0 and RM 1.0,[86] some of the peak cPAH concentrations were at depth, and some were in the uppermost interval; there were no notable concentration gradients within these cores. Peak cPAH concentrations in these cores ranged from 140 to 590 µg/kg dw, except for one core (LDW-SC11), located between RM 0.5 and RM 0.6 on the western shoreline, which had a peak cPAH concentration of 4,400 µg/kg dw in the uppermost interval (0 to 0.8 ft). The deeper intervals of this core did not have detectable cPAH concentrations. At two sampling locations near RM 0.6 (LDW-SC12 and DR044), the cPAH concentration was higher in the co-located surface sediment sample (1,600 µg/kg dw) that was associated with both of these subsurface samples.

In summary, cPAH concentrations in surface and subsurface sediment in this section of the LDW were variable; many subsurface cores had cPAH concentrations greater than the area-based 75th percentile. In Slip 1, high cPAH concentrations were detected primarily at depth within the cores. At other locations with high cPAH concentrations (i.e., greater than the area-based 95th percentile), peak concentrations were in the uppermost intervals at four locations and at depth at two locations.

RM 1.0 to RM 2.2

Concentrations of cPAHs were variable in both surface and subsurface sediment in this section of the LDW. Along the eastern shoreline between RM 1.0 and RM 1.5, subsurface cPAH concentrations were relatively low (≤ 610 µg/kg dw) in cores that were sectioned into more than one subsurface interval, with the exception of location LDW-SC23, which had a high cPAH concentration (4,600 µg/kg dw) in the 3-to-3.5-ft interval (Map 4-41b). The core from location LDW-SC18 near RM 1.0 had a peak cPAH

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[86] The remaining cores were located at RM 0.35 on the western shoreline (one core), near RM 0.5 on the western shoreline (one core), in the channel (three cores), north of Kellogg Island (two cores), and south of Kellogg Island (one core).

concentration in subsurface sediment of 510 µg/kg dw in the 0-to-1-ft interval, compared with a relatively high cPAH concentration in the co-located surface sediment sample (3,000 µg/kg dw).

Along the western shoreline between RM 1.0 and RM 1.3, two cores were collected; peak cPAH concentrations (500 µg/kg dw in both cores) were detected at depth (2 to 4 ft) in the core near RM 1.0 and in the uppermost interval in the core near RM 1.2.

Four cores collected between RM 1.3 and RM 1.4 on the west side of the navigation channel had high cPAH concentrations at depth (Map 4-41b). In two of these cores (LDW-SC25 and DR054), the peak cPAH concentrations (980 and 2,000 mg/kg dw, respectively) were in the 2-to-4-ft interval. In the other two cores (LDW-SC26 and LDW-SC28), the peak cPAH concentrations (4,000 and 1,400 µg/kg dw, respectively) were within the approximate interval of 6 to 8 ft. Between RM 1.4 and RM 1.5, four cores were sampled at only one interval each, with concentrations ranging from 280 to 620 µg/kg dw; this area was dredged after the cores were collected. In the only other core collected between RM 1.3 and RM 1.5 (LDW-SC29), cPAH concentrations were relatively low (≤ 150 µg/kg dw).

Between RM 1.5 and RM 1.7 on the east side of the navigation channel in an area that has been dredged, cPAH concentrations in subsurface sediment were generally less than the 75th percentile, and most locations were sampled at only one relatively large interval (generally greater than 3 ft). Therefore, peak cPAH concentrations could not be determined at these locations. Similarly, in the navigation channel, subsurface cPAH concentrations were less than the 75th percentile, and samples were collected at only one relatively large interval of 4 ft.

The subsurface sample with the highest cPAH concentration in the entire LDW (7,000 µg/kg dw) was collected from the head of Slip 3, in the 1-to-2-ft interval. The concentration in the uppermost interval (0 to 1 ft) was lower (2,800 µg/kg dw), as were surface sediment concentrations based on IDW interpolation (≤ 1,100 µg/kg dw).

cPAH concentrations in the remaining nine core locations on the east and west sides of the navigation channel between RM 1.7 and RM 2.2 were ≤ 750 µg/kg dw. Peak concentrations were at depth, except at locations LDW-SC39 and LDW-SC36, which had relatively low peak concentrations (260 and 110 µg/kg dw, respectively) in the uppermost interval.

In summary, in this section of the LDW, peak cPAH concentrations were detected at depth at all of the locations that had high subsurface cPAH concentrations (i.e., greater than the 95th percentile), and at most of the locations that had cPAH concentrations greater than the 75th percentile.

RM 2.2 to RM 3.0

In the section of the LDW between RM 2.2 and RM 3.0, cPAH concentrations were generally variable, with concentrations between the 25th and 95th percentile in both surface and subsurface sediment (Map 4-41c). The highest concentrations in both surface and subsurface sediment were located in Slip 4 and near RM 2.7 on the west shoreline.

In Slip 4, subsurface cPAH concentrations ranged from 280 to 3,400 µg/kg dw in four cores sampled over relatively large intervals (ranging from the surface to approximately 3 to 4 ft); thus, the depth of peak cPAH concentrations could not be determined. This area was subsequently dredged. Just downstream from Slip 4, a peak cPAH concentration of 1,000 µg/kg dw was detected at depth (in the 2-to-4-ft interval of core LDW-SC45).

Four cores were collected on the east side of the navigation channel between RM 2.2 and RM 2.7. cPAH concentrations were very low in one core (≤ 20 µg/kg dw at LDW-SC43). In the other three cores, peak cPAH concentrations ranged from 120 to 550 µg/kg dw; peak cPAH concentrations were detected at depth within each core.

In five cores collected within the navigation channel, cPAH concentrations were ≤ 290 µg/kg dw. Samples were collected from one interval in four of these cores. In the remaining core (DR171), cPAH concentrations were similar (250 to 290 µg/kg dw) in the uppermost intervals and at depth.

On the west side of the navigation channel, the highest subsurface cPAH concentrations were detected between RM 2.6 and RM 2.8. In four of the five cores collected in this area, only one interval (approximately 0 to 3 ft) was sampled in each core; cPAH concentrations in these cores ranged from 340 to 2,800 µg/kg dw. The area was dredged after these cores were collected. The remaining core (LDW-SC46) was collected after dredging. The peak cPAH concentration (1,200 µg/kg dw) at this location was detected at depth (in the 1-to-2-ft interval). However, the cPAH concentration was highest (3,100 µg/kg dw) in the co-located surface sediment sample at this location.

The remaining five cores on the west side of the navigation channel between RM 2.2 and RM 2.6 had subsurface cPAH concentrations of ≤ 570 µg/kg dw. Only two of these cores were sampled at more than one interval. The peak cPAH concentration was at depth in core DR137 (290 µg/kg dw in the 2-to-4-ft interval) and in the uppermost interval in core LDW-SC40 (51 µg/kg dw in the 0-to-1.3-ft interval).

In summary, all of the cores sampled at more than one interval and with cPAH concentrations ≥ 290 µg/kg dw had peak cPAH concentrations at depth. The location of the peak cPAH concentration is unknown in many cores collected between RM 2.2 and RM 3.0 because they were sampled at only one interval.

RM 3.0 to RM 4.0

In general, cPAH concentrations in surface and subsurface sediment between RM 3.0 and RM 4.0 were less than or equal to the area-based 50th percentile (260 µg/kg dw) (Map 4-41d). Subsurface cPAH concentrations were ≤ 460 µg/kg dw at all locations except two. Of the eight locations east of the navigation channel (including four locations within the Boeing Plant 2/Jorgensen Forge EAA), peak concentrations were in the uppermost interval at three locations: near RM 3.3 (1,500 µg/kg dw at SD-04107), near RM 3.8 (2,200 µg/kg dw at LDW-SC51), and near RM 3.8 (360 µg/kg dw at LDW-SC50a). At the other five locations on the east side of the navigation channel, subsurface cPAH concentrations ranged from 18 to 460 µg/kg dw. Some of the peak cPAH concentrations were at depth, and some were in the uppermost interval at these locations. Two of the eight locations east of the navigation channel (LDW-SC50a and SB-04117) had cPAH concentrations in co-located surface sediment samples (1,800 and 2,500 µg/kg dw, respectively) that were at least 25% greater than those in subsurface intervals.

Eight cores were collected within the navigation channel between RM 3.0 and RM 4.0. Subsurface cPAH concentrations in these cores ranged from 33 to 270 µg/kg dw. The locations of peak cPAH concentrations in six of these cores are unknown because samples were collected from only one interval (either 0 to 2 ft or 0 to 4 ft). At the remaining two locations, the peak cPAH concentration of 40 µg/kg dw was detected within the uppermost interval (0 to 1 ft) of core LDW-SC48 and the peak concentration of 270 µg/kg dw was detected at depth (1-to-2-ft interval) in core LDW-SC49.

On the west side of the navigation channel, cPAH concentrations ranged from 18 to 340 µg/kg dw in cores collected from 12 locations. At two of these locations near RM 3.4, cPAHs were not detected in samples collected from the 0-to-4-ft interval. At five locations between RM 3.5 and RM 3.7 (in the T-117 EAA), there were no data for the uppermost intervals; cPAH concentrations in the samples collected at depth were ≤ 97.5 µg/kg dw. At two of the locations in the T-117 EAA (T117-SE-25-SC and T117-SE-37-SC), cPAH concentrations were much higher in co-located surface sediment samples (1,200 and 11,000 µg/kg dw, respectively) than in subsurface intervals. At the remaining five locations, where samples were collected at more than one interval, peak cPAH concentrations were ≤ 340 µg/kg dw and were detected at depth in four cores and in the uppermost interval in one core (Map 4-41d).

In summary, cPAH concentrations in surface and subsurface sediment were generally less than the 50th percentile in this section of the LDW. At the two locations with the highest cPAH concentrations in subsurface sediment (2,200 and 1,500 µg/kg dw at LDW-SC51 and SD-04107, respectively), peak concentrations were in the uppermost interval within the cores. At other locations where more than one interval was sampled, some peak concentrations were at depth, and some were in the uppermost interval.

RM 4.0 to RM 5.0

Most of the cPAH concentrations in both surface and subsurface sediment between RM 4.0 and RM 5.0 were less than the 50th percentile (230 µg/kg dw), except in and around Slip 6 and upstream of RM 4.6 (Map 4-41e).

In Slip 6, surface and subsurface cPAH concentrations were generally highest toward the head of the slip. Seventeen of the 19 subsurface sediment cores within the slip and downstream from the slip on the east side of the navigation channel were sampled at relatively shallow single intervals (either 0.33 to 0.69 ft or 0.33 to 0.82 ft); therefore, insufficient data were available to evaluate the depth of peak cPAH concentrations at these locations. cPAH concentrations in the eight shallow samples from within the slip ranged from 630 µg/kg dw at the mouth of the slip (SB-8) to 2,100 µg/kg dw at the head of the slip (SB-17). Some locations in Slip 6 had higher cPAH concentrations in co-located surface sediment samples than in associated subsurface cores, but only three of the locations (SB-12, SB-3, and SB-5) had concentrations in surface sediment that were both greater than the 75th percentile and at least 25% greater than concentrations in subsurface sediment. cPAH concentrations were generally lower downstream of the slip, ranging from 310 to 880 µg/kg dw. In one of the cores with available data from the head of the slip (LDW-SC53), the peak cPAH concentration (1,200 µg/kg dw) was detected in the uppermost interval (0 to 2 ft). In the other core with available data (DR246), just outside the slip, concentrations were similar in the two intervals (230 and 210 µg/kg dw in the 0-to-2-ft and 2-to-4-ft intervals, respectively).

All cores collected in the navigation channel between RM 4.0 and RM 5.0 were sampled at one interval and had relatively low cPAH concentrations, ranging from 14 to 170 µg/kg dw, except for one. This core (from location "Average of 10-12") had a peak cPAH concentration (1,100 µg/kg dw) in the uppermost interval (0 to 4 ft) and a lower cPAH concentration (57 µg/kg dw) at depth (4-to-11-ft interval).

Ten cores were collected along the west side of the LDW between RM 4.0 and RM 5.0. cPAH concentrations in these cores were less than or equal to the 50th percentile (230 µg/kg dw). Six of these cores were sampled at one interval (approximately 0 to 2 ft) in an area that was subsequently dredged near RM 4.1 (Map 4-41e). In the remaining four cores with available data, three cores had peak cPAH concentrations in the uppermost interval, and one had a peak cPAH concentration at depth; all cPAH concentrations were less than the 50th percentile.

Four cores were collected between RM 4.8 and RM 5.0, three of which were within the Norfolk EAA. The core collected outside the EAA had no detected cPAH concentrations. In the two cores within the EAA (NFK009 and NFK008) and one core just outside the EAA (NFK207), peak concentrations (220, 270, and 1,460 µg/kg dw, respectively) were all in the uppermost interval of the core, within the 0-to-1-ft interval. The area within the Norfolk EAA was dredged and backfilled with clean material after the two cores at NFK009 and NFK008 were collected. The interpolated surface sediment data shown on Map 4-41e reflect post-dredging conditions.

In summary, most of the subsurface cPAH concentrations between RM 4.0 and RM 5.0 were less than the 50th percentile, except in Slip 6 and in the Norfolk EAA, which had concentrations up to 2,100 and 1,460 µg/kg dw, respectively. Of the 11 cores sampled at more than one interval in this section of the LDW, all but one had the peak concentration in the uppermost interval.

4.2.5.4 Tissue

The tissue dataset for the RI contained cPAH data for 214 composite samples of fish, shellfish, and benthic invertebrates; all of these samples were analyzed for each of the seven individual cPAH compounds. Table 4-56 presents a summary of cPAH concentrations for each tissue type. At least one cPAH compound was detected in 65% of tissue samples. The highest cPAH concentrations were detected in invertebrates (clams, mussels, and benthic invertebrates), with mean concentrations ranging from 15 to 44 µg/kg ww. cPAH concentrations in individual composite samples of all other species were less than 3 µg/kg ww, with the exception one Pacific staghorn sculpin sample with a concentration of 36 µg/kg ww from subarea T3. cPAHs were not detected in any of the other 23 Pacific staghorn sculpin samples, although the RLs for these samples (36 to 360 µg/kg ww) were elevated because analytical dilutions were used to minimize matrix interferences. The Pacific staghorn sculpin samples were analyzed by the same low-level analytical method, EPA 8270-SIM, as the other tissue samples collected in 2004. RLs for historical samples were generally higher than for samples collected in 2004; this uncertainty was discussed in Section B.6.1.1.5 of the HHRA.

Table 4-56. Summary of cPAH data in tissue

	Detection Frequency		Concentration (µg/kg ww)[a]			
Tissue Type	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	Non-Detect or Range of Non-Detects[c]
Fish, Whole Body						
English sole	18/21	86	0.45 J	2.8	1.6	0.45
Pacific staghorn sculpin	1/24	4	36 J	36 J	130	36 – 360
Shiner surfperch	24/27	89	0.37 J	2.2	3.1	43
Starry flounder	3/3	100	0.47 J	0.66 J	0.59	na
Fish, Fillet						
English sole (with skin)	4/7	57	0.37 J	0.53	0.35	0.45
English sole (without skin)	0/6	0	nd	nd	8.9	6.4 – 29
Pile perch (with skin)	1/1	100	0.43 J	0.43 J	0.43	na
Starry flounder (with skin)	1/1	100	0.64 J	0.64 J	0.64	na
Striped perch (with skin)	1/1	100	0.43 J	0.43 J	0.43	na
Crab, Edible Meat						
Dungeness crab	6/9	67	0.54 J	0.84 J	3.7	0.65 – 29
Slender crab	2/12	17	0.33	0.63 J	0.35	0.65

	DETECTION FREQUENCY		CONCENTRATION (µg/kg ww)[a]			
TISSUE TYPE	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	CALCULATED MEAN[b]	NON-DETECT OR RANGE OF NON-DETECTS[c]
Crab, Hepatopancreas						
Dungeness crab	2/4	50	0.73 J	0.73 J	6.5	6.2 – 43
Slender crab	4/4	100	0.68 J	0.71	0.70	na
Crab, Whole Body (calc'd)[d]						
Dungeness crab	nc	nc	0.60 JM	2.4 JM	2.6	17
Slender crab	nc	nc	0.45 M	0.67 M	0.65	na
Invertebrates, Whole Body						
Amphipod	0/4	0	nd	nd	22	43
Benthic invertebrates	20/20	100	4.2 J	290 J	44	na
Shellfish						
Mussels, wild (edible meat)	11/22	50	30	33	23	29
Mussels, transplanted (edible meat)	16/34	47	29	35	27	29 – 50
Soft-shell clam (whole body)	14/14	100	6.8	44	15	na

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

[b] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[c] Non-detect values were calculated for samples with no detected cPAH compounds. Non-detect values represent the sum of one-half the RL multiplied by the PEF for each compound.

[d] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. cPAH concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
J – estimated concentration
M – calculated whole-body concentration
na – not applicable
nc – not calculated
nd – not detected
PEF – potency equivalency factor
RL – reporting limit
ww – wet weight

Mean cPAH concentrations by sampling area for the various tissue types and sediment SWACs for the tissue sampling areas are presented on Map 4-43. cPAH concentrations in whole-body English sole samples were significantly lower in Area T4 than in Areas T1 and T2. For shiner surfperch, mean concentrations were significantly lower in Area T4 than in Area T3. In Dungeness crab samples, cPAH concentrations were similar among all areas, except the single sample collected from Area T4 had a higher concentration. The relationships between cPAH concentrations in fish or crabs and sediment were not statistically evaluated. However, cPAH concentrations in English sole and shiner surfperch tissue and sediment (SWAC) were lowest in Area T4 (Map 4-43). There is no apparent relationship between cPAH concentrations in crab tissue and sediment in the four areas based on the data shown on Map 4-43.

All 14 clam composite tissue samples had detected concentrations of cPAHs. The maximum cPAH concentration in clams (44 µg/kg ww) was detected at location C8 in Slip 4, which had the highest cPAH concentration in sediment co-located with clam tissue samples (7,100 µg/kg ww) (Map 4-43). A detailed evaluation of the cPAH concentrations in clam tissue and co-located sediment did not find a strong relationship (see Section 8.3.3).

The primary component of the cPAH concentration in tissue was benzo(a)pyrene, which accounted for 50% of the carcinogenic potency of PAHs in tissue on average (standard deviation of 10%). Benzo(b)fluoranthene accounted for an average of 20% of the carcinogenic potency of PAHs (standard deviation of 10%), and the remaining five individual cPAHs each accounted for 7% or less of the carcinogenic potency of PAHs.

4.2.5.5 Surface water

Surface water samples collected by King County as part of the KC WQA (King County 1999a) did not contain detectable concentrations of any individual cPAH compounds. Because many organic compounds are not typically detected in ambient grab samples, particularly in the dissolved phase, King County also deployed SPMDs to collect data for selected organic compounds, including PAHs, at 1 and 3 m below the water surface near the Duwamish/Diagonal (RM 0.5) and Brandon (RM 1.1) CSO outfalls, as described in Section 4.1.2.4. Ambient PAH concentrations in water were calculated using the mass of chemicals detected in each SPMD and the chemical-specific partition coefficients between polyethylene and water, which were derived from laboratory experiments conducted at Battelle's Marine Science Laboratory.

Calculated cPAH concentrations from SPMD samples ranged from 0.000055 to 0.00010 µg/L (Table 4-57). Calculated benzo(a)pyrene and benzo(a)anthracene concentrations accounted for an average of 36% and 32%, respectively, of the total carcinogenic potency of PAHs among the four SPMD samples. The remaining five individual cPAH compounds each accounted for an average of 10% or less of the total carcinogenic potency of PAHs. Individual cPAH concentrations in surface water samples are compared with available WQC in Section 4.2.2 (Table 4-20).

Table 4-57. cPAH concentrations calculated from SPMDs deployed near the Duwamish/Diagonal and Brandon CSOs

CSO	CALCULATED cPAH CONCENTRATION (µg/L)[a]	
	1-m Depth	3-m Depth
Duwamish/Diagonal	0.00010	0.000071
Brandon	0.000068	0.000055

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

cPAH – carcinogenic polycyclic aromatic hydrocarbon

CSO – combined sewer overflow
LDW – Lower Duwamish Waterway
SPMD – semi-permeable membrane device

4.2.5.6 Summary of cPAH data

cPAH concentrations in surface sediment ranged from 9.7 to 11,000 µg/kg dw; the area-based 95th and 50th percentiles calculated from the RI baseline dataset were 1,100 and 230 µg/kg dw, respectively. The highest cPAH concentrations in surface sediment were in the T-117 EAA, in the Slip 4 EAA, in the RM 3.8 EAA on the eastern shoreline, near RM 0.6 on the western shoreline, near RM 0.0 close to Harbor Island, and in the Duwamish/Diagonal EAA.

Areas with the highest cPAH concentrations in subsurface sediment were Slip 3, near RM 1.2 on the east side, near RM 0.5 on the western shoreline, and near RM 1.4 west of the navigation channel. Peak cPAH concentrations were generally detected at depth within cores at locations with more than one interval sampled and with concentrations greater than the area-based 95th percentile (1,100 µg/kg dw). Cores with peak cPAH concentrations detected in uppermost intervals were located at the head of Slip 6, in the Norfolk EAA, near RM 0.5 on the western shoreline, and in some of the cores from the Duwamish/Diagonal EAA.

The highest cPAH concentrations in tissue were detected in clams, mussels, and benthic invertebrates, with mean concentrations ranging from 15 to 44 µg/kg ww. cPAH concentrations detected in individual composite samples of all other species were less than 3 µg/kg ww, with the exception of one Pacific staghorn sculpin sample with a concentration of 36 µg/kg ww. Although low-level analytical methods were used for all tissue samples collected in 2004, RLs were high for Pacific staghorn sculpin samples because analytical dilutions were used to minimize matrix interferences at the discretion of the laboratory for the sake of reducing impact to their analytical equipment. The dilutions resulted in reduced sensitivity of the analysis (i.e., higher RLs) for this subset of samples.

Mean cPAH concentrations in English sole (whole-body tissue) were significantly lower in Area T4 than in Areas T1 and T2. For shiner surfperch, mean concentrations were significantly lower in Area T4 than in Area T3. The relationship between cPAH concentrations in clam tissue and sediment was relatively uncertain (Section 8.3.3).

Grab surface water samples did not contain detectable concentrations of individual cPAH compounds. Concentrations of cPAHs calculated from SPMD samples ranged from 0.000055 to 0.00010 µg/L.

4.2.6 BEHP

This section summarizes the nature and extent of BEHP concentrations in surface sediment, subsurface sediment, tissue, surface water, seep water, and porewater. BEHP is a risk driver chemical for benthic invertebrates; detected concentrations of

BEHP exceeded the SQS in 5.8% and the CSL in 7.1% of surface sediment samples, second only to the number of samples with PCB exceedances.

4.2.6.1 Surface sediment

Surface sediment samples from 832 of the 1,365 locations in the RI baseline dataset were analyzed for BEHP. BEHP was detected in 81% of surface sediment samples analyzed for BEHP, with concentrations ranging from 5.4 to 14,000 µg/kg dw (Table 4-58). The numerical 95th percentile based on BEHP concentrations in the surface sediment dataset was 2,400 µg/kg dw (Table 4-59). The area-based 95th percentile calculated using Theissen polygons (990 µg/kg dw) was less than half of the numerical 95th percentile, because many of the surface sediment samples were collected in more contaminated areas, and are thus not representative of area-wide conditions.

Table 4-58. Summary of surface sediment data for BEHP

Detection Frequency			Concentration					
Ratio	%	Unit	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]	SQS	CSL
674/832	81	µg/kg dw	5.4	14,000	600	15 – 1,500	na	na
637/764	83	mg/kg OC	0.20	387	25	0.88 – 170	47	78

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] RLs are based only on non-detect samples.

BEHP – bis(2-ethylhexyl) phthalate
CSL – cleanup screening level
dw – dry weight
na – not applicable
OC – organic carbon
RL – reporting limit
SQS – sediment quality standards

Table 4-59. Percentiles of BEHP concentrations in LDW surface sediment collected from RM 0.0 to RM 6.0

	BEHP Concentration (µg/kg dw)				
Method	Entire Dataset	25th Percentile	50th Percentile	75th Percentile	95th Percentile
Area-based	380 (SWAC)	84	200	440	990
Numerical	600 (mean)	95	230	490	2,400

BEHP – bis(2-ethylhexyl) phthalate
dw – dry weight
LDW – Lower Duwamish Waterway
RM – river mile
SWAC – spatially weighted average concentration

The majority of locations with the highest BEHP concentrations in surface sediment were between RM 0.4 and RM 0.6 in and near the Duwamish/Diagonal EAA.[87] Other areas with more than one sample with BEHP concentrations greater than the area- based 75th percentile were near RM 2.2 on the west side of the LDW (EAA 2), in Slip 4, and near RM 3.5 on the east side (in the Boeing Plant 2/Jorgensen Forge EAA) (Map 4-44).

BEHP concentrations exceeded the SQS but not the CSL at 48 locations (5.8%) and exceeded the CSL at 59 locations (7.1%) (Table 4-16 in Section 4.2.1). At the remainder of the locations (approximately 87%), BEHP concentrations were less than the SQS (Figure 4-27). Most of the BEHP SQS and CSL exceedances were located within EAAs, particularly between RM 0.4 and RM 0.6, in and around the Duwamish/Diagonal EAA (Map 4-44).

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[87] Monitoring data for BEHP concentrations as a function of year are shown for the Duwamish/Diagonal and Norfolk in Appendix I, Sections I.4.1 and I.4.8, respectively.

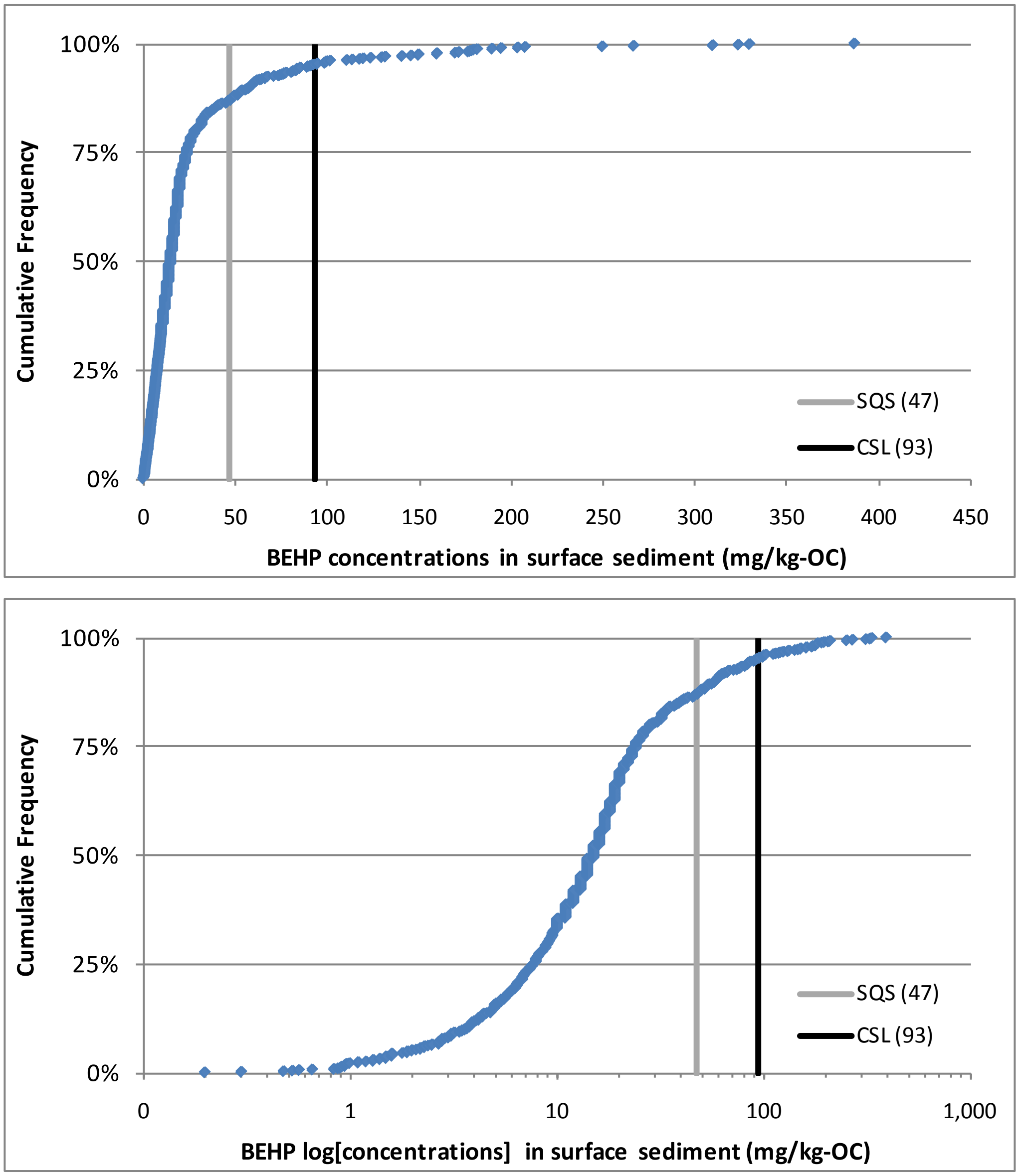

Figure 4-27. Cumulative frequency of OC-normalized BEHP concentrations in surface sediment (arithmetic and log-scale)

An analysis was conducted to assess changes in BEHP concentrations at 44 locations sampled in the 1990s that were resampled during sampling events in 2004 through 2006; such comparisons were limited to those locations where sediments were sampled within 10 ft of the original sample and where BEHP was detected in either the original or the later sample (Table 4-60).[88]

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[88] The older data within 10 ft of the newer sample were excluded from the baseline dataset as described in Section 4.1.2.1.

Table 4-60. Difference in BEHP concentrations at resampled locations sorted by concentration difference

River Mile	Location ID		Date		Concentration (µg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
0	DR076	LDW-SS5	8/24/1998	1/17/2005	6,100	20 U	-6,080
1.3	DR053	LDW-SS44	8/31/1998	1/21/2005	3,800 J	120	-3,680
1.4	SS-4	LDW-SS55	8/17/1993	1/24/2005	2,200	98	-2,102
1.4	DR160	LDW-SS51	8/12/1998	1/18/2005	1,900	120 U	-1,780
4.2	R40	LDW-SS127	10/13/1997	1/20/2005	1,400	140	-1,260
3.8	DR187	LDW-SS115	8/27/1998	1/25/2005	1,500	330	-1,170
4.2	R45	LDW-SS130	10/16/1997	1/20/2005	1,200	72	-1,128
1.7	DR097	LDW-SS63	8/20/1998	1/21/2005	1,200	150	-1,050
0.3	DR079	LDW-SS15	8/24/1998	1/17/2005	1,100	64 U	-1,036
1.4	DR030	LDW-SS50	8/17/1998	1/24/2005	1,500	560	-940
0.3	DUD042	LDW-SS17	11/11/1995	1/24/2005	2,000	1,100	-900
1.3	SS-2	LDW-SS48	8/17/1993	1/18/2005	1,600	770	-830
1.4	SS-3	LDW-SS49	8/17/1993	1/26/2005	960	160	-800
3.8	R24	LDW-SS117	10/10/1997	1/20/2005	940	140	-800
4.2	R42	LDW-SS129	10/13/1997	1/20/2005	930	170	-760
2.2	DR113	LDW-SS81	8/19/1998	3/8/2005	910	190 U	-720
0.1	K-07	LDW-SS4	9/30/1991	1/17/2005	740 J	83 U	-657
1.0	DR019	LDW-SS32	8/17/1998	1/18/2005	710	93 U	-617
0.2	DR035	LDW-SS12	8/11/1998	1/17/2005	720	180	-540
2.1	DR106	LDW-SS76	8/19/1998	1/20/2005	460	59	-401
1.0	DR020	LDW-SS31	8/17/1998	1/21/2005	550	160	-390
1.4	DR065	LDW-SS52	8/17/1998	1/25/2005	410	95	-315
1.5	DR123	LDW-SS57	9/14/1998	1/24/2005	560	290	-270
1.4	DR028	B4b	8/17/1998	8/28/2004	390	140 J	-250
3.7	SD-323-S	LDW-SS110	8/17/2004	1/25/2005	410	170	-240
2.8	DR175	LDW-SS94	8/20/1998	1/21/2005	270	46	-224
0.9	DR021	LDW-SS319	8/17/1998	10/4/2006	710	520	-190
3.9	R30	LDW-SS119	10/11/1997	1/19/2005	460	280	-180
1.2	DR088	LDW-SS40	8/31/1998	1/18/2005	410	270	-140
4.3	DR286	B10b	8/26/1998	8/19/2004	150 UJ	35 J	-115
2.0	R7	LDW-SS75	10/15/1997	1/21/2005	180	74	-106
4.7	DR271	LDW-SS148	9/15/1998	3/9/2005	260	160 U	-100
3.2	DR202	LDW-SS104	8/27/1998	1/25/2005	80	36 J	-44
4.1	DR238	LDW-SS125	8/27/1998	1/20/2005	130	97 J	**-33**
3.1	DR198	LDW-SS102	8/20/1998	1/24/2005	150 UJ	130	**-20**

River Mile	Location ID		Date		Concentration (µg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
3.7	R21	LDW-SS113b	10/9/1997	1/20/2005	220	200	**-20**
0.9	DR085	LDW-SSB2b	8/31/1998	3/11/2005	340	350	**10**
4.1	A11-05	LDW-SS126	8/18/1994	1/20/2005	81	92 J	**11**
1.0	DR087	LDW-SS37	8/12/1998	1/18/2005	570	760	190
1.9	DR131	LDW-SS70	8/13/1998	1/21/2005	1,500	1,700	**200**
3.7	DR186	LDW-SS111	8/27/1998	1/19/2005	210	580	370
4.2	DR242	SB-1	8/24/1998	8/25/2004	620	1,600	980

Note: Concentrations in bold were within the range of analytical variability (≤ 25% increase or decrease compared with the initial concentration).

BEHP – bis(2-ethylhexyl) phthalate

dw – dry weight

ID – identification

J – estimated concentration

U – not detected at RL shown

UJ – not detected at estimated RL shown

At the 44 resampled surface sediment locations, BEHP concentrations decreased over time at 33 locations (79%), increased at 3 locations (7%), and were indistinguishable from the initial concentration at 6 locations (14%)[89] (Table 4-60 and Map 4-45). Two resampled locations were not included in Table 4-60 because BEHP was not detected in either of the paired samples; RLs ranged from 67 to 710 µg/kg dw. At the three locations with higher concentrations in more recent samples, differences ranged from 190 to 980 µg/kg dw, with the greatest increase in a sample from the head of Slip 6. Concentration differences ranged from 44 to 6,080 µg/kg dw at the 35 locations with lower concentrations in more recent samples, with the greatest decrease located near RM 0.0. [90]

To determine if BEHP concentrations were significantly different at locations that were resampled (i.e., if the mean difference between the paired concentrations was greater than zero), a non-parametric, paired t-test (Wilcoxon signed-rank test) was used. The test was run separately for each reach of the LDW (as defined in the STM) and for the entire LDW. Data were not included for 10 locations in Table 4-60 that were sampled less than 6 years apart or where BEHP was not detected in either the initial or later sample.

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[89] Concentration differences at these locations were within the range of analytical variability (≤ 25% increase or decrease compared to the initial concentration).

[90] Information on heterogeneity of BEHP in sediment related to sample homogenization and laboratory variability is provided by field duplicate samples (i.e., sample splits from a grab sample after homogenization). The relative percent difference in BEPH concentrations between paired field duplicate samples in the RI (n = 7) ranged from 0 to 38%, with an average of 18%.

When the data were analyzed by reach of the LDW, the differences in between initial and resampled BEHP concentrations were significant at the 95% confidence level in the reach between RM 0.0 and RM 2.2 (Table 4-61; Figure 4-28). When data were analyzed for the entire LDW, differences between the initial and resampled BEHP concentrations were significant at the 95% level.

Table 4-61. Results of statistical analyses of differences in BEHP concentrations in surface sediment at resampled locations

REACH OF THE LDW	n[a]	**MEAN CONCENTRATION (µg/kg dw)**		**SIGNIFICANCE (p value)**
		INITIAL	**RESAMPLED**	
RM 0.0 to RM 2.2	19	1,100	400	**0.001**
RM 2.2 to RM 4.0	7	530	230	0.13
RM 4.0 to RM 4.8	6	730	360	0.25
Entire LDW	32	880	350	**< 0.005**

[a] Data were not included for 10 locations (DR076, DR160, DR079, DR113, K-07, DR019, SD-323-S, DR286, DR271, and DR198) because they were sampled less than 6 years apart or BEHP was not detected in either the initial or later sample.

BEHP – bis(2-ethylhexyl) phthalate

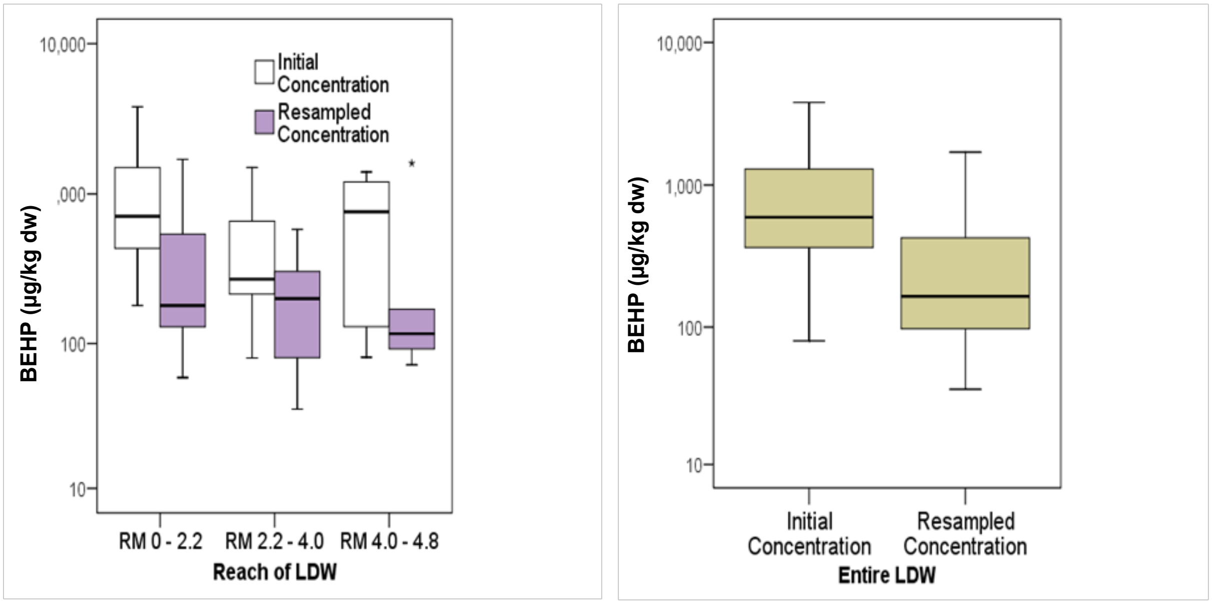
dw – dry weight

LDW – Lower Duwamish Waterway

n – number of samples

RM – river mile

Bold identifies values that are significant at the 95% confidence level.



Note: The boxes indicate the 25th and 75th percentiles within which 50% of the data lie. The length of this box indicates the interquartile range (IQR). The line in the center of the box is the 50th percentile (median). The whiskers of the box plot indicate the maximum and minimum values from the dataset that are within 1.5 IQRs of the 25th and 75th percentile values. Outliers are indicated by a circle and have a value between 1.5 and 3 of the IQR from the upper or lower edge of the box (75th or 25th percentile). Extreme values are indicated by an asterisk and have a value > 3 times the IQR from the 75th or 25th percentile. Data were not included for 10 locations because they were sampled less than 6 years apart or BEHP was not detected in either the initial or later sample.

Figure 4-28. Differences in BEHP concentrations in surface sediment at resampled locations

4.2.6.2 Subsurface sediment

Subsurface sediment cores were sampled at various intervals during different sampling events. To calculate summary statistics, data were categorized according to specified intervals, as described for total PCBs in Section 4.2.3.2.

BEHP was detected in 71% of the subsurface samples in which phthalates were analyzed, at concentrations ranging from 12 to 5,100 µg/kg dw (Table 4-62). Maps 4-46a through 4-46c present BEHP data in 1- and 2-ft subsurface intervals for the top 6 ft of sediment. Most of the locations with BEHP concentrations in subsurface sediment that were greater than the 95th percentile were located between RM 0.4 and RM 0.6, in the vicinity of the Duwamish/Diagonal EAA. The maximum concentration in this area was 18,000 µg/kg dw in the 3-to-6-ft interval at DUD261.

Table 4-62. Summary of subsurface sediment data for BEHP

SAMPLING INTERVAL (ft)	DETECTION FREQUENCY		CONCENTRATION (µg/kg dw)		
	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLs[a]
All data					
Any interval	216/306	71	12 J	5,100	19 – 2,400
1-ft intervals[b]					
0 to 1	44/64	69	22	2,100	27 – 1,400
1 to 2	24/45	53	13 J	3,900	19 – 400
2 to 3	7/10	70	20 J	820	20 – 59
3 to 4	2/9	22	17 J	24	19 – 20
4 to 5	1/1	100	290	290	na
2-ft intervals[b]					
0 to 2	65/74	88	12 J	2,400	20 – 530
2 to 4	51/73	70	13 J	3,900	19 – 280
4 to 6	8/10	80	56 J	2,200	20.0 – 65
6 to 8	4/5	80	1,000	3,800	65
8 to 10	2/5	40	72.00	260	61 – 66

Note: Data included in the subsurface dataset are described in Section 4.1.2.2; subsurface sediment samples collected from areas that were subsequently dredged are not summarized in this table because they do not represent existing conditions.

[a] RLs were based only on non-detect samples.

[b] For the calculation of summary statistics, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to the 1- or 2-ft sampling category that best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were averaged to obtain a calculated concentration for the corresponding 2-ft interval. Some of the subsurface data were collected from intervals other than 1 or 2 ft; therefore, the detection frequency ratio for "any interval" may reflect data for intervals not included in these categories.

BEHP – bis(2-ethylhexyl) phthalate
dw – dry weight
J – estimated concentration
na – not applicable
RL – reporting limit

4.2.6.3 Surface and subsurface sediment patterns for BEHP

BEHP concentrations in surface and subsurface sediment were evaluated to determine whether peak concentrations were detected in the uppermost interval of sediment cores or at depth. The specific depths of peak concentrations within sediment cores were also evaluated to assess patterns (Maps 4-47a through 4-47e). The percentiles of BEHP concentrations in sediment discussed in this section were calculated on an area basis using the method described in the introduction to Section 4.2 and Thiessen polygon data. Area-based percentiles are discussed rather than numerical percentiles because they are more representative of area-wide conditions. In calculating numerical percentiles, each sampling location is given equal weight although more samples were collected in contaminated areas. As discussed in Section 4.2.3.3 for total PCBs, although there is some uncertainty in the interpretation of patterns and spatial relationships between BEHP concentrations in surface and subsurface data, this analysis provides one line of evidence to understand the conceptual site model for BEHP in the LDW. Specific surface sediment locations were noted in the discussion if: 1) BEHP concentrations were higher in surface sediment samples than in sampling intervals of co-located subsurface cores, 2) the BEHP concentration in surface sediment was greater than the 75th percentile, and 3) the BEHP concentration in surface sediment was at least 25% greater than the highest arsenic concentration in any subsurface interval. BEHP concentrations in all co-located surface and subsurface sediment samples are presented on Maps 4-47a through 4-47e. Maps 4-48a and 4-48b show only the subsurface sediment locations with co-located surface sediment data.

RM 0.0 to RM 1.0

Between RM 0.0 and RM 1.0, BEHP concentrations in surface sedment greater than the 95th percentile of 990 µg/kg dw were located within or near the Duwamish/Diagonal EAA (Map 4-47a). In subsurface sediment, BEHP concentrations greater than the 95th percentile were located in the Duwamish/Diagonal EAA and also near Harbor Island and Slip 1.

Six cores were collected north of RM 0.3. Two of these cores had relatively low or non-detected BEHP concentrations in subsurface sediment (Map 4-47a; LDW-SC3 and LDW-SC5). The remaining four cores (LDW-SC1, LDW-SC2, LDW-SC4, and LDW-SC6) had peak BEHP concentrations ranging from 830 to 2,400 µg/kg dw at depth within the cores.

Of the 13 subsurface cores collected within the Duwamish/Diagonal EAA at more than one depth interval, all but three (DUD254, DUD006, and DUD261) had the peak BEHP concentration in the uppermost interval (generally 0 to 3 ft). Identifying the depth of the peak BEHP concentration within the EAA is generally not possible because most of the cores had relatively large sampling intervals of 3 ft. Surface sediment data were also evaluated for four locations within the Duwamish/Diagonal EAA where surface sediment samples were collected within 10 ft of the subsurface cores. At two of these locations (DR008 and DUD027), concentrations in surface sediment (11,000 and 13,000 µg/kg dw, respectively) were at least 25% higher than those in associated subsurface sediment cores (Map 4-47a).

Of the four subsurface cores collected just outside the Duwamish/Diagonal EAA at more than one depth interval, two had peak BEHP concentrations in the uppermost intervals (1,200 µg/kg dw in the 0-to-1-ft interval at LDW-SC7 and 1,700 µg/kg dw in the 0-to-1-ft interval at LDW-SC9). These two core locations had higher BEHP concentrations in co-located surface sediment samples (2,600 µg/kg dw at LDW-SC7 and 2,420 µg/kg dw at LDW-SC9).The other two cores had peak BEHP concentrations at depth (2,200 µg/kg dw in the 4-to-6-ft interval at LDW-SC8 and 3,900 µg/kg dw in the 2-to-4-ft interval at LDW-SC10).

Four subsurface cores were collected within Slip 1 (LDW-SC15, LDW-SC16, LDW-SC17, and DR021). The peak BEHP concentrations were located within the 2-to-4-ft interval in each of these cores, with concentrations ranging from 480 to 3,100 µg/kg dw. One subsurface core located just downstream from Slip 1 (LDW-SC13) had low concentrations of BEHP (≤ 160 µg/kg dw).

In one subsurface core located near RM 0.9 in the navigation channel (LDW-SC14), the peak BEHP concentration of 1,200 µg/kg dw was in the uppermost core interval, although the concentration in a co-located surface sediment sample was lower (350 µg/kg dw).

The remaining four subsurface cores were collected near RM 0.5 along the western shoreline (LDW-SC11), north of Kellogg Island near RM 0.6 (LDW-SC12 and DR044), and south of Kellogg Island near RM 1.0 (LDW-SC19). BEHP concentrations in these cores were relatively low (≤ 380 µg/kg dw) and peak BEHP concentrations were at depth in three of the four cores. The one core with a peak concentration in the uppermost interval was LDW-SC11 (310 µg/kg dw in the 0-to-1-ft interval).

In summary, outside of the Duwamish/Diagonal EAA, the majority of subsurface core samples with BEHP concentrations greater than the 75th percentile had peak concentrations at depth within the cores. Within the Duwamish/Diagonal EAA, most of the peak BEHP concentrations were in the uppermost depth intervals of the cores, although this interval was generally large (i.e., greater than 3 ft). Surface sediment data also indicated that peak BEHP concentrations may be highest within the surface intervals in most locations in this area.

RM 1.0 to RM 2.2

In this reach of the LDW, BEHP concentrations in both surface and subsurface sediment were variable, and generally less than the 95th percentile (990 µg/kg dw). Between RM 1.0 and RM 1.3, seven subsurface cores were sectioned into more than one depth interval. In three of these cores, BEHP concentrations were relatively low (≤ 390 µg/kg dw). In three other cores, the peak BEHP concentrations were between the 75th and 95th percentiles (440 and 990 µg/kg dw, respectively) and were detected at depth within the cores (at 2 to 4 ft in cores from LDW-SC21 and DR025, and at 3.5 to 4 ft in the core from LDW-SC23). In the remaining core (LDW-SC20), collected from the navigation channel at RM 1.0, the highest BEHP concentration was in the surface interval of the core (620 µg/kg dw at 0 to 2 ft) and was higher in a surface sediment sample (760 µg/kg dw) collected at the same location (Map 4-47b).

Between RM 1.3 and 1.5, six cores collected on both the east and west side of the navigation channelwere sectioned into more than one depth interval. Peak BEHP concentrations in all but one of these cores were relatively high, ranging from 740 to 3,900 µg/kg dw. In three of the cores, the peak concentrations were at depth (580 µg/kg dw at the 2-to-4-ft interval in LDW-SC25, 3,900 µg/kg dw at the 6-to-8-ft interval in LDW-SC26, and 1,000 µg/kg dw at the 5.5to-7.5-ft interval in LDW-SC28).

In another two of the five cores (DR054 and LDW-SC27), the peak concentrations (1,200 and 910 µg/kg dw, respectively) were in the uppermost interval (0 to 2 ft), although the concentrations were lower in co-located surface sediment samples (450 and 560 µg/kg dw, respectively) (Map 4-47b). In the remaining core collected between RM 1.3 and RM 1.5 (LDW-SC29), BEHP concentrations were below the 25th percentile of 84 µg/kg dw.

Between RM 1.5 and RM 1.7 on the east side of the navigation channel in an area that has been dredged, BEHP concentrations in subsurface sediment were generally less than the 75th percentile, and most locations were sampled at only one relatively large interval (generally greater than 3 ft). Therefore, the depth of peak BEHP concentrations could not be determined at these locations. In the navigation channel between RM 1.8 and RM 2.2, subsurface BEHP concentrations were relatively low (less than the 75th percentile of 440 µg/kg dw), and samples were collected at only one relatively large interval of 4 ft.

In Slip 2, one core was collected within the slip (LDW-SC32), and one core was collected at the mouth (DR101). In both cores, BEHP concentrations were greater than the 75th percentile and were detected at depth (650 µg/kg dw at the 1-to-2-ft interval and 1,400 µg/kg dw at the 2-to-4-ft interval).

At the remaining 10 subsurface locations sampled between RM 1.8 and RM 2.2, all but 3 locations had peak BEHP concentrations less than the 75th percentile. At the three locations with concentrations greater than the 75th percentile (LDW-SC34, LDW-SC37, and DR112), peak concentrations were at depth within the cores.

In summary, the depth of the peak BEHP concentration is unknown in most cores collected within the navigation channel and in many of the cores collected from the east side of the navigation channel between RM 1.5 and RM 1.8. In 12 of the 15 cores with BEHP concentrations greater than the 75th percentile, peak concentrations were at depth within the core.

RM 2.2 to RM 3.0

Between RM 2.2 and RM 3.0, BEHP concentrations in both surface and subsurface sediment were variable and generally less than the 75th percentile Map 4-47c). Two subsurface cores were collected in the vicinity of EAA 2, where three surface sediment samples had high BEHP concentrations (ranging from 2,300 to 4,200 µg/kg dw). The peak BEHP concentration of 450 µg/kg dw in one of the two cores (DR137) was in the 2-to-4 ft interval, and BEHP concentrations were very low (less than or equal to 84 µg/kg dw) in the other core (LDW-SC40).

Between RM 2.4 and RM 2.8 on the west side, nine of the ten subsurface cores were sampled at only one depth interval, so the depth of the peak BEHP concentration within those cores could not be determined. In the one core sampled at more than one depth interval (LDW-SC46 near RM 2.7), the BEHP concentrations were similar throughout the core (200 to 250 µg/kg dw) and were slightly higher in the surface sediment sample collected at the same location (430 µg/kg dw).

Five subsurface cores were collected within the navigation channel. Only one of these cores (DR171) was analyzed at more than one depth interval. The peak BEHP concentration (460 µg/kg dw) in this core was in the 2-to-4-ft interval.

Between RM 2.3 and RM 2.8 on the east side of the navigation channel, five subsurface sediment cores were collected as part of the RI. The core from LDW-SC41 was the only core with BEHP concentrations greater than 220 µg/kg dw. In this core, the peak concentration of 480 µg/kg dw was in the uppermost interval of 0 to 1 ft.

Slip 4 had high BEHP concentrations (up to 5,100 µg/kg dw) in surface sediment samples collected at the head of the slip, but no subsurface samples were collected from this area. Subsurface samples collected from the western half of the slip were analyzed at only one depth interval; BEHP concentrations in these cores were relatively low (< 300 µg/kg dw).

In summary, the depth of the peak BEHP concentration is unknown in many of the cores collected between RM 2.2 and RM 3.0 because they were sampled at only one depth interval. Most subsurface BEHP concentrations were less than the 75th percentile. In three of the cores with BEHP concentrations greater than the 75th percentile, two had peak concentrations at depth, and one had a peak concentration in the uppermost interval.

RM 3.0 to RM 4.0

Most of the BEHP concentrations in surface sediment between RM 3.0 and RM 4.0 were less than the 75th percentile, and most concentrations in subsurface sediment were less than the 50th percentile (Map 4-47d). In subsurface cores from locations west of the navigation channel and outside of the T-117 EAA, BEHP concentrations were generally low. In the only two cores with concentrations greater than the 50th percentile, one peak BEHP concentration was at depth (350 µg/kg dw in the 2-to-3-ft interval at SC47) and one peak concentration was in the uppermost interval (220 µg/kg dw in the 0-to-2.5-ft interval at SB-5).

Eight subsurface cores were collected within the navigation channel. Six of these cores were analyzed at only one depth interval. The remaining two cores had BEHP concentrations less than the 25th percentile of 84.

In the Boeing Plant 2/Jorgensen Forge EAA, five subsurface cores were collected. Each of these cores was either analyzed at only one depth interval, or not all depth intervals were analyzed, so it is not possible to determine the depth of peak BEHP concentrations.

In the T-117 EAA, six subsurface cores were collected. Five of these cores were either analyzed at only one interval or not all depth intervals were analyzed. In the one core with data from two depth intervals (DR206), the peak BEHP concentration (520 µg/kg dw) was in the uppermost interval (0-to-2 ft). The BEHP concentration was lower (260 µg/kg dw) in the co-located surface sediment sample.

Between RM 3.8 and 3.9 on the east side of the navigation channel, two subsurface cores had peak BEHP concentrations at depth and two had peak concentrations in the uppermost intervals. In cores from LDW-SC50a and LDW-SC52, peak BEHP concentrations (680 and 660 µg/kg dw, respectively) were in the 0-to-1-ft interval. At sampling location LDW-SC50a, the BEHP concentration (1,200 µg/kg dw) in the co-located surface sediment sample was higher than the peak concentration in the subsurface core. In cores from LDW-SC51 and DR220, peak concentrations (1,800 and 470 µg/kg dw, respectively) were in the 0.5-to-1-ft and 2-to-4-ft intervals, respectively.

In summary, BEHP concentrations in surface and subsurface sediment were generally less than the 75th percentile in this section of the LDW. At the five locations where BEHP concentrations in subsurface cores were greater than the 75th percentile and where data were available for more than one interval, three cores had peak concentrations in the uppermost interval and two had peak concentrations at depth. In addition, one core location had data from only one subsurface depth interval but had a peak BEHP concentration in the co-located surface sediment sample.

RM 4.0 to RM 5.0

Concentrations of BEHP in surface and subsurface sediment between RM 4.0 and RM 5.0 were variable, with most concentrations less than the 75th percentile. In the six subsurface cores collected from a dredged area on the west side of the navigation channel between RM 4.0 and RM 4.2, BEHP concentrations were low, ranging from 160 to 340 µg/kg dw. Only one interval was sampled in each of these cores, so the depth of the peak BEHP concentration could not be determined. In one subsurface core collected just outside this dredged area (DR284), the peak BEHP concentration (650 µg/kg dw) was at depth (2 to 4 ft).

Three subsurface cores were collected between RM 4.2 and RM 4.8 on the west side of the navigation channel. BEHP concentrations were very low in these cores (less than or equal to 140 µg/kg dw).

At the Rhône-Poulenc site between RM 4.0 and RM 4.3 on the east side of the navigation channel (including Slip 6), 20 very shallow subsurface core samples were collected and only one subsurface interval (0.33 to approximately 0.7 ft) was analyzed from each. In nine of these cores, surface intervals (0 to 0.33 ft) were also analyzed for BEHP; these samples were classified as surface sediment samples in the RI. The surface and subsurface intervals in most of these nine cores had very similar BEHP concentrations (Map 4-47e). All of the subsurface cores in this area with BEHP concentrations greater than the 95th percentile were in cores collected within the slip. Two deeper subsurface cores were collected in the area near Rhône-Poulenc: LDW-SC53 near the head of the slip and DR246 near the mouth of the slip. Peak BEHP concentrations (880 and 650 µg/kg dw, respectively) were at depth (2 to 4 ft) in both of these cores.

In the Norfolk EAA, three subsurface cores were collected. Peak BEHP concentrations were in the uppermost interval (0-to-1 ft) in each of these cores (600 µg/kg dw at NFK008, 1,400 µg/kg dw at NFK207 and 570 µg/kg dw at NFK009). It should be noted that the area where these three cores were collected was subsequently dredged and capped; surface sediment data at these locations represent post dredging/capping conditions. At one location just outside the Norfolk EAA (LDW-SC55), BEHP was not detected at any depth interval.

In summary, most of the very shallow subsurface cores collected from the vicinity of Slip 6 had similar BEHP concentrations in the 0-to-0.33-ft and 0.33-to-0.7-ft intervals. In the Norfolk EAA, peak BEHP concentrations were greater than the 75th percentile, and were in the uppermost intervals (0-to-1 ft); cores in this area were collected before dredging and capping. Three other subsurface cores between RM 4.0 and RM 5.0 had concentrations greater than the 75th percentile; peak concentrations in these cores were at depth.

4.2.6.4 Tissue

BEHP data are available for 227 composite samples of fish, shellfish, and benthic invertebrate tissues (Table 4-63). BEHP was detected most frequently in soft-shell clams (in 10 of 14 samples); other samples of various tissue types generally had low detection frequencies of BEHP (Table 4-63). The highest concentrations of BEHP were detected in a benthic invertebrate composite tissue sample from LDW B1b near RM 0.1 (2,200 µg/kg ww) and a shiner surfperch sample collected from Area T2E (2,100 µg/kg ww).

Table 4-63. Summary of BEHP data in specific tissue types

Tissue Type	Detection Frequency		Concentration (µg/kg ww)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Fish, Whole Body					
English sole	1/27	4	170	170	50 – 3,600
Pacific staghorn sculpin	0/24	0	nd	nd	490 – 5,000
Shiner surfperch	5/34	15	280 J	2,100 J	24 – 3,600
Starry flounder	0/3	0	nd	nd	66 – 67
Fish, Fillet					
English sole (with skin)	2/7	29	1,100	1,300 J	67 – 130
English sole (without skin)	0/6	0	nd	nd	3.6 – 16
Pile perch (with skin)	0/1	0	nd	nd	67
Starry flounder (with skin)	0/1	0	nd	nd	67

TISSUE TYPE	DETECTION FREQUENCY		CONCENTRATION (µg/kg ww)		
	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLs[a]
Striped perch (with skin)	0/1	0	nd	nd	67
Crab, Edible Meat					
Dungeness crab	0/9	0	nd	nd	16 – 130
Slender crab	0/12	0	nd	nd	66 – 260
Crab, Hepatopancreas					
Dungeness crab	0/4	0	nd	nd	24 – 130
Slender crab	1/4	25	100 J	100 J	66 – 230
Crab, Whole Body (calc'd)[b]					
Dungeness crab	nc	nc	nd	nd	9 – 55
Slender crab	nc	nc	100 JM	100 JM	33 – 100
Invertebrates, Whole Body					
Amphipod	2/4	50	170	530	24
Benthic invertebrates	5/20	25	1,100 J	2,200 J	2,500
Shellfish					
Mussels, wild (edible meat)	2/22	9	28	190	16
Mussels, transplanted (edible meat)	0/34	0	nd	nd	16 – 26
Soft-shell clam (whole body)	10/14	71	56 J	220 J	490 – 500

[a] RLs are based only on non-detect samples.

[b] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

BEHP – bis(2-ethylhexyl) phthalate
J – estimated concentration
M – calculated whole-body concentration
na – not applicable
nc – not calculated
nd – not detected
RL – reporting limit
ww – wet weight

BEHP RLs were elevated (up to 5,000 µg/kg ww) in some of the English sole, Pacific staghorn sculpin, shiner surfperch, and benthic invertebrate samples collected for the RI in 2004 because of analytical dilutions. Original RLs in 49 tissue samples of various types were very high (7,200 µg/kg ww), resulting in the need for re-analysis using additional sample extract clean-up steps and alternate analytical methods to achieve lower RLs. All 49 samples with original RLs of 7,200 µg/kg ww were reanalyzed; 71 other samples with elevated RLs ranging from 240 to 5,000 µg/kg ww were not reanalyzed. The remaining 22 samples collected for the RI in 2004 had detected concentrations and were also not reanalyzed. BEHP was detected in only one of the re-analyzed tissue samples at a concentration of 100 µg/kg ww; RLs for the re-analyses of the remaining 48 samples ranged from 66 to 230 µg/kg ww. BEHP was detected in tissue only once in the 2006 King County sampling event in Area T1 (170 µg/kg ww in an English sole composite sample) (Anchor and King County 2007). RLs for the 2006 King County analyses of English sole and shiner surfperch from Area T1 ranged from 50 to 200 µg/kg ww. There were 74 samples collected between 1992 and 1998 in the RI dataset. BEHP was detected in four of these samples at concentrations of 28 to 530 µg/kg ww, and RLs ranged from 3.6 to 26 µg/kg ww. The reanalysis of the subset of 2004 tissues suggests that RLs in the 2004 dataset would have been much lower if samples had not been diluted; the more recent results from 2006 suggest that BEHP is rarely detected in fish.

4.2.6.5 Surface water

On multiple occasions, surface water samples were collected in the LDW along transects near three CSO outfalls and analyzed for SVOCs as part of the King County WQA (King County 1999e). BEHP detection frequencies ranged from 18 to 31% (Table 4-64), with concentrations ranging from 0.14 to 23.8 µg/L. Phthalates were often detected in the method blank samples, and therefore, these results were likely influenced by laboratory contamination. During data validation, detected sample concentrations were compared with the detected method blank concentrations, and sample concentrations less than 10 times the concentration in the associated method blank samples were qualified as non-detected at elevated RLs for 67 results. Concentrations of BEHP in surface water samples are compared with WQC in Section 4.2.2 (Table 4-20).

Table 4-64. Summary of surface water data for BEHP at three locations in the LDW

	Detection Frequency		Concentration (µg/L)		
Location ID[a]	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
BRN (RM 1.1)	7/39	18	0.21	3.51	0.14 – 1.31
SWM (RM 1.9)	8/42	19	0.302	23.8	0.14 – 0.667
NFK (RM 4.9)	4/13	31	0.14	0.253	0.15 – 0.859

[a] BRN, SWM, and NFK are located in the LDW near the Brandon, SW Michigan, and Norfolk CSOs, respectively.

[b] RLs are based only on non-detect samples.

BEHP – bis(2-ethylhexyl) phthalate
CSO – combined sewer overflow
ID – identification
LDW – Lower Duwamish Waterway
RL – reporting limit
RM – river mile

4.2.6.6 Seep water

BEHP was analyzed in 3 seep water samples collected at T-117; 7 seep water samples collected at Rhône-Poulenc in 1995; 9 and 5 seep water samples collected at GWI in 1998 and 1999, respectively; 17 seep water samples collected at Boeing Plant 2; and 16 seep water samples collected throughout the LDW as part of the RI in 2004 (Map 4-11a).

BEHP was detected in two unfiltered seep water samples from the Rhône-Poulenc sampling event and one unfiltered seep water sample from the T-117 sampling event, at concentrations ranging from 8.9 to 27.0 µg/L (Table 4-65; Maps 4-11d and 4-11e). BEHP was not detected in any of the 15 filtered seep water samples or 16 unfiltered samples collected throughout the LDW as part of the RI sampling event. BEHP was not detected in the filtered seep water samples, so comparisons to WQC were not made in Section 4.2.2.

Table 4-65. Summary of seep water data for BEHP

SAMPLING EVENT/YEAR	DETECTION FREQUENCY		DETECTED CONCENTRATION (µg/L)[a]	RL (µg/L)[b]
	RATIO	%		
Rhône-Poulenc RFI-3 (1995)	2/7	29	13.8 – 27.0	nr[c]
T-117 (2003)	1/3	33	8.9 J	1.0

[a] Unfiltered samples

[b] RLs are based only on non-detect samples.

[c] RLs for non-detected chemicals were not reported for this sampling event.

BEHP – bis(2-ethylhexyl) phthalate
J – estimated concentration
nr – not reported
RFI – RCRA facility investigation
RL – reporting limit
T-117 – Terminal 117

4.2.6.7 Porewater

The only porewater samples from the LDW that have been analyzed for BEHP were those collected during the 2004 Rhône-Poulenc sampling event. BEHP was detected at locations SH-02, SH-03, SH-04, SH-05, and SH-08 at concentrations up to 390 µg/L (Table 4-66; Map 4-12d).[91] BEHP concentrations in porewater samples are compared with WQC in Section 4.2.2 (Table 4-22).

Table 4-66. Summary of BEHP concentrations in porewater samples collected from Rhône-Poulenc

DETECTION FREQUENCY		CONCENTRATION (µg/L)		
RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLS[a]
5/9	56	2.0	390	1.9

[a] RLs are based only on non-detect samples.

BEHP – bis(2-ethylhexyl) phthalate
J – estimated concentration
RL – reporting limit

4.2.6.8 Summary of BEHP data

BEHP concentrations in surface sediment ranged from 5.4 to 14,000 µg/kg dw; the area-based 95th and 50th percentiles calculated from the RI baseline dataset were

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[91] BEHP was not detected in two method blanks at RLs of 2 µg/L.

990 and 440 µg/kg dw, respectively. The highest BEHP concentrations in both surface and subsurface sediment were between RM 0.4 and RM 0.6 in and near the Duwamish/Diagonal EAA. Peak BEHP concentrations in subsurface sediment were frequently detected at depth within cores at concentrations greater than the area-based 75th percentile (440 µg/kg dw), except for the Duwamish/Diagonal, which had relatively large depth intervals, and the Norfolk EAA, where dredging was conducted after cores were collected.

In tissue, BEHP was detected most frequently in soft-shell clams; only 17 of the other 227 samples of various tissue types had detected concentrations of BEHP. The highest concentrations of BEHP were detected in a benthic invertebrate composite tissue sample from LDW B1b near RM 0.1 (2,200 µg/kg ww) and a shiner surfperch sample collected from Area T2E (2,100 µg/kg ww). BEHP RLs for some of the English sole, Pacific staghorn sculpin, shiner surfperch, and benthic invertebrate samples collected for the RI in 2004 were elevated (up to 7,200 µg/kg ww) because of analytical dilutions. Reanalysis of 49 tissue samples of various types with RLs greater than 5,000 µg/kg ww resulted in substantially lower RLs and only one detected concentration. In addition, more recent results from 2006 suggested that BEHP is rarely detected in fish.

In surface water samples collected from three LDW locations, detection frequencies of BEHP ranged from 18 to 31%, with concentrations ranging from 0.14 to 23.8 µg/L. BEHP was detected in unfiltered seep water samples collected from two of the four site-specific sampling events (Rhône-Poulenc and T-117) at concentrations ranging from 8.9 to 27 µg/L in unfiltered samples. BEHP was not detected in any of the 15 filtered seep water samples or 16 unfiltered samples collected throughout the LDW during the RI sampling event. Rhône-Poulenc is the only area from which porewater samples have been collected and analyzed for BEHP. During the 2004 Rhône-Poulenc sampling event, BEHP was detected in porewater at five locations at concentrations ranging from 2 to 390 µg/L.

4.2.7 Dioxins and furans

This section summarizes the nature and extent of dioxin and furan concentrations in surface and subsurface sediment samples. Dioxins and furans have not been analyzed in tissue, surface water, seep water, or porewater samples collected from the LDW.

4.2.7.1 Surface sediment

Surface sediment samples from within the LDW area were analyzed for dioxin and furan congeners from 26 locations as part of the RI (Windward 2005d) and from 28 locations as part of the EPA SI (Weston 1999a). At least one dioxin or furan congener was detected in every sample, with individual congener concentrations ranging from 0.0590 to 241,000 ng/kg dw (Table 4-67).

Table 4-67. Summary of dioxin and furan data in LDW surface sediment

	Detection Frequency		Concentration (ng/kg dw)			
Chemical	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	Non-Detect or Range of Non-Detects[b]
2,3,7,8-TCDD	26/54	48	0.0660 J	30.6	1.5	0.27 – 1.1
1,2,3,7,8-PeCDD	27/54	50	0.100 J	57.1	4.2	0.53 – 4.1
1,2,3,4,7,8-HxCDD	27/54	50	0.193 J	124	8.2	0.72 – 5.4
1,2,3,6,7,8-HxCDD	45/54	83	0.978 J	3,400	100	0.74 – 4.3
1,2,3,7,8,9-HxCDD	38/54	70	0.537 J	315	25	0.537 – 11.3
1,2,3,4,6,7,8-HpCDD	52/54	96	25.5	73,700	2,500	0.99 – 1.1
OCDD	54/54	100	7.8 J	241,000	16,000	na
2,3,7,8-TCDF	44/54	81	0.113 J	397	12	0.18 – 1.7
1,2,3,7,8-PeCDF	26/54	48	0.0950 J	69.3	4.5	0.28 – 5.0
2,3,4,7,8-PeCDF	27/54	50	0.212	230	15	0.44 – 5.4
1,2,3,4,7,8-HxCDF	39/54	72	0.513 J	2,530	120	0.29 – 4.2
1,2,3,6,7,8-HxCDF	27/54	50	0.174 J	365	21	0.22 – 4.3
1,2,3,7,8,9-HxCDF	25/54	46	0.0730 J	33.8 J	1.8	0.12 – 2.4
2,3,4,6,7,8-HxCDF	27/54	50	0.155 J	302 J	12	0.29 – 2.5
1,2,3,4,6,7,8-HpCDF	51/54	94	5.18 J	40,300	1,000	0.62 – 7.7
1,2,3,4,7,8,9-HpCDF	36/54	67	0.385 J	3,720	100	0.77 – 4.2
OCDF	53/54	98	12.5	93,700	2,600	0.74
Dioxin and furan mammalian TEQ (Van den Berg et al. 2006)[c]	54/54	100	0.878	2,100 J	82	na

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] Non-detect values were calculated for samples with no detected dioxin or furan congeners. Non-detect values represent the sum of one-half the RL multiplied by the TEF for each compound.

[c] The UCL for dioxin and furan TEQs in surface sediment was 610 ng/kg dw using the RI baseline dataset as calculated in Appendix C (Table C.3-8). This UCL is presented as the EPC for the netfishing scenario in the HHRA (Appendix B), which uses the entire surface sediment dataset.

dw – dry weight
EPC – exposure point concentration
HHRA – human health risk assessment
HpCDD – heptachlorodibenzo-*p*-dioxin
HpCDF – heptachlorodibenzofuran
HxCDD – hexachlorodibenzo-*p*-dioxin
HxCDF – hexachlorodibenzofuran
J – estimated concentration
LDW – Lower Duwamish Waterway
na – not applicable
OCDD – octachlorodibenzo-*p*-dioxin
OCDF – octachlorodibenzofuran
PeCDD – pentachlorodibenzo-*p*-dioxin
PeCDF – pentachlorodibenzofuran
RL – reporting limit
TCDD – tetrachlorodibenzo-*p*-dioxin
TCDF – tetrachlorodibenzofuran
TEF – toxic equivalency factor
TEQ – toxic equivalent
UCL – upper confidence limit on the mean
WHO – World Health Organization

TEQs were calculated using mammalian TEFs for 17 dioxin or furan congeners.[92] Methods for these calculations are described in Appendix B, Section B.2.2.4. Dioxin and furan TEQs in surface sediment ranged from 0.878 to 2,100 ng/kg dw (Table 4-67). Forty-two of the fifty-four dioxin and furan TEQs were less than 20 ng/kg dw. The median and 95th percentile TEQs were 10 and 490 ng/kg dw, respectively (Figure 4-29).

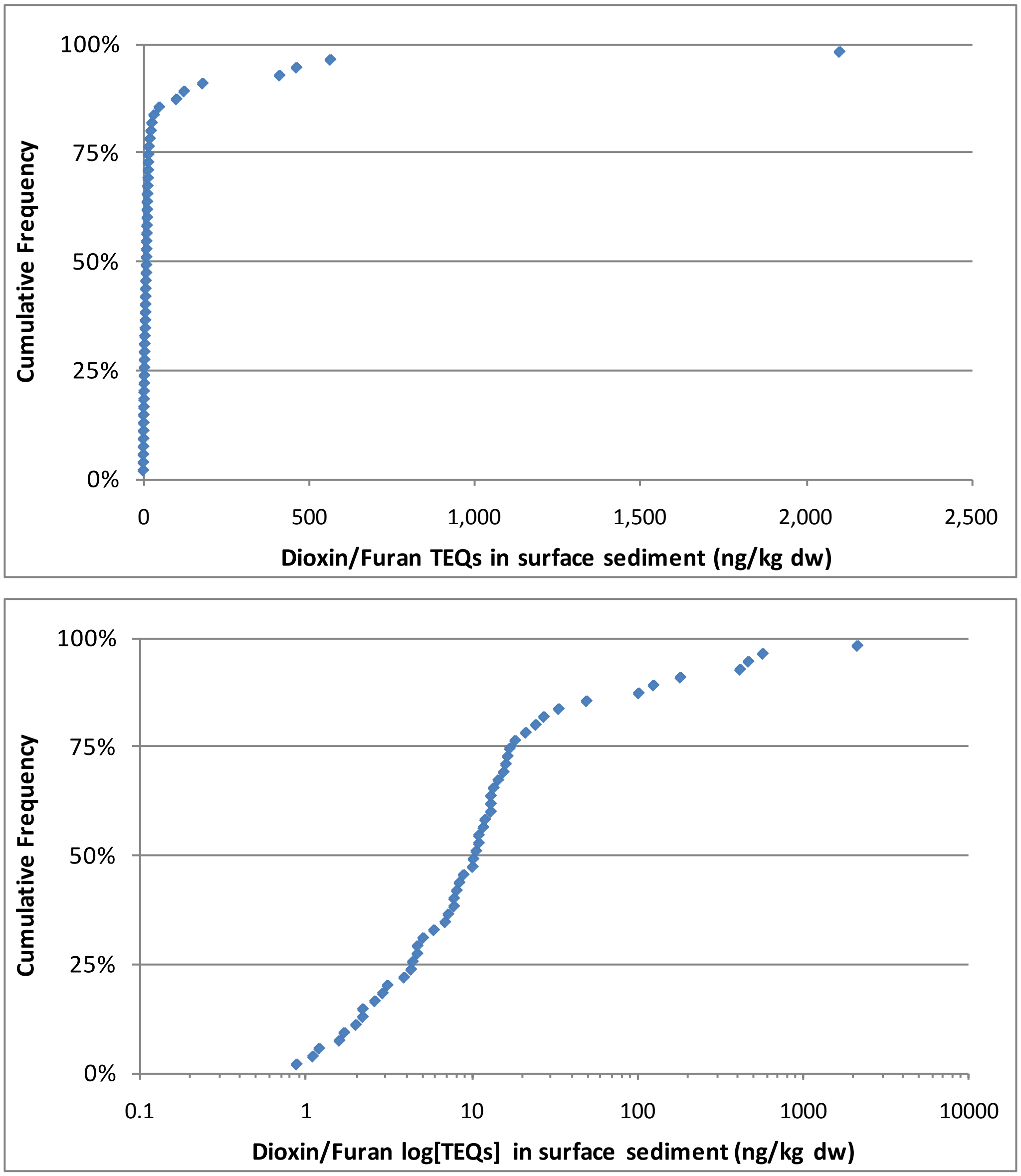

Figure 4-29. Cumulative frequency of dioxin and furan TEQs in surface sediment (arithmetic and log-scale)

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[92] For the purposes of describing the distribution of dioxins and furans in the LDW, TEQ concentrations were calculated using TEFs for mammals as presented in Van den Berg et al. (2006). If a dioxin or furan congener concentration was reported as undetected, then the TEF was multiplied by half the RL.

The three highest dioxin and furan TEQs (2,100, 565, and 463 ng/kg dw) were detected in surface sediment samples collected from the embayment between RM 1.4 and RM 1.5 on the west side of the LDW (Map 4-49). Other areas with notable dioxin and furan TEQs were: 1) the inlet at RM 2.2 in EAA 2 on the west side of the LDW (412 ng/kg dw), 2) near RM 0.5 E in the Duwamish/Diagonal EAA (180 ng/kg dw), 3) near RM 1.0 in the navigation channel (124 ng/kg dw), and 4) near RM 3.6 E in the Boeing Plant 2/Jorgensen Forge EAA (101 ng/kg dw).

Seven samples analyzed for dioxins and furans were not included in the baseline RI dataset but were included in Table 4-67 and Map 4-49 at the request of EPA and Ecology.[93] Four of these samples were collected during the EPA SI (DR021 at RM 0.9, DR065 at RM 1.4, DR187 at RM 3.8, and DR238 at RM 4.1) and were not included in the baseline RI dataset because older data were excluded if a location was resampled within 10 ft (see Section 4.1.2.1). However, dioxins and furans were not analyzed in the more recent samples from these four EPA SI locations. Dioxin and furan TEQs in the four EPA SI samples ranged from 3.1 to 13 ng/kg dw. The remaining three samples not included in the baseline RI dataset were collected during the RI (LDW-SS18 and LDW-SS20 at RM 0.4, and LDW-SS22 at RM 0.5) and were excluded because of their proximity to the Duwamish/Diagonal EAA dredge perimeter; the baseline dataset was intended to only represent pre-dredge conditions (see Section 4.1.2.1). Dioxin and furan TEQs in these three RI samples ranged from 0.878 to 24.2 ng/kg dw.

4.2.7.2 Subsurface sediment

Subsurface sediment samples were analyzed for dioxins and furans from eight locations as part of the RI. These locations were selected because of relatively high dioxin and furan TEQs in nearby surface sediment. Dioxin and furan TEQs at these eight subsurface sediment locations ranged from 0.147 to 194 ng/kg dw (Table 4-68; Map 4-49). The highest TEQ (194 ng/kg dw) was detected in the 4-to-6-ft interval in the core from location LDW-SC20 in the navigation channel at RM 1.0.

Table 4-68. Summary of dioxin and furan mammalian TEQ data in LDW subsurface sediment

SUBSURFACE SEDIMENT			SURFACE SEDIMENT	
SUBSURFACE CORE SAMPLING LOCATION	SAMPLING INTERVAL (ft)	TEQ (ng/kg dw)[a]	SAMPLING LOCATION CLOSEST TO SUBSURFACE CORE	TEQ (ng/kg dw)[a]
LDW-SC19 (RM 1.0 south of Kellogg Island)	0 – 1	22.8 J	LDW-SS36	27.1 J
	1 – 2	20.1 J		
	2 – 4	20.5 J		

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[93] The entire surface sediment dataset, including data that were not included in the baseline RI dataset, are available on a CD that accompanies this RI.

SUBSURFACE SEDIMENT			SURFACE SEDIMENT	
SUBSURFACE CORE SAMPLING LOCATION	SAMPLING INTERVAL (ft)	TEQ (ng/kg dw)[a]	SAMPLING LOCATION CLOSEST TO SUBSURFACE CORE	TEQ (ng/kg dw)[a]
LDW-SC20 (RM 1.0 in the navigation channel)	0 – 2	38.7 J	LDW-SS37	124 J
	2 – 4	27.1		
	4 – 6	194 J		
	8 – 10	5.60 J		
LDW-SC26 (RM 1.4 on the west side of the navigation channel)	0 – 1	15.9 J	none[b]	no data
	1 – 2	13.1 J		
	2 – 4	22.4 J		
	6 – 8	136 J		
LDW-SC28 (RM 1.4 on the west side of the navigation channel)	0 – 1	19.9 J	none[b]	no data
	1 – 2	14.8		
	2 – 4	18.5 J		
LDW-SC29 (RM 1.4 west of the Glacier Northwest dock)	0 – 1	54.1 J	LDW-SS56	2,100 J
	1 – 2	1.03 J		
	2 – 4	0.147 J		
LDW-SC39 (between RM 2.1 and RM 2.2 on the western shoreline)	0 – 1	7.91 J	none[b]	no data
	1 – 2	12.4 J		
	2 – 4	13.1 J		
LDW-SC40 (near RM 2.3 on the western shoreline)	0 – 1	6.71 J	none[b]	no data
	1 – 2	0.485 J		
	2 – 4	0.355 J		
LDW-SC41 (near RM 2.4 on the east side in the Myrtle Street Embayment)	0 – 1	13.8	LDW-SS83	33.3 J
	1 – 2	12.5 J		
	2 – 4	14.0 J		

[a] TEQs were calculated using TEFs for mammals presented in Van den Berg et al. (2006).

[b] No surface samples were collected near subsurface cores.

dw – dry weight
J – estimated concentration
LDW – Lower Duwamish Waterway
RM – river mile
TEQ – toxic equivalent

Core LDW-SC29, collected near the surface sediment location with the highest TEQ (2,100 ng/kg dw) had low TEQs in comparison (Table 4-68). The highest TEQ in this subsurface core was 54.1 ng/kg dw in the 0-to-1-ft interval. In core LDW-SC26 collected just outside the embayment, the highest TEQ (136 ng/kg dw) was detected at an interval of 6 to 8 ft. A third subsurface core collected in that general area (LDW-SC28) had relatively low TEQs (14.8 to 19.9 ng/kg dw) in all three subsurface intervals.

In core LDW-SC20 near RM 1.0 in the navigation channel, the highest TEQ (194 ng/kg dw in the 4-to-6-ft interval) was higher than the TEQ in the surface sediment sample from that area (124 ng/kg dw).

In the other two cores collected from locations where surface sediment data were available (LDW-SC19 and LDW-SC41), TEQs in subsurface samples were slightly lower than TEQs in associated surface sediment samples (up to 22.8 ng/kg dw in subsurface samples and up to 33.3 ng/kg dw in surface samples) (Table 4-68). There was little variability in TEQs among the different subsurface intervals in each core.

4.2.8 Metals

This section summarizes the nature and extent of metals and TBT concentrations in surface sediment, subsurface sediment, tissue, surface water, seeps, and porewater.

4.2.8.1 Surface sediment

Surface sediment samples collected from 884 locations were analyzed for at least one metal or TBT. Table 4-69 presents detection frequencies and a summary of concentrations in surface sediment for nine metals (cadmium, chromium, copper, lead, mercury, nickel, silver, vanadium, and zinc) and TBT.[94]

Table 4-69. Summary of surface sediment data for metals and TBT

	Detection Frequency		Concentration (mg/kg dw)				SMS Criteria (mg/kg dw)	
Chemical	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]	SQS	CSL
Cadmium	584/838	70	0.030 J	120	1	0.04 – 2.5	5.1	6.7
Chromium	850/850	100	4.8	1,100 J	40	na	260	270
Copper	852/852	100	5	12,000 J	100	na	390	390
Lead	852/852	100	2	23,000	100	na	450	530
Mercury	746/868	86	0.021	4.6 J	0.2	0.02 – 0.10	0.41	0.59
Nickel	812/812	100	5	910	30	na	nv	nv
Silver	499/823	61	0.020	270	1	0.046 – 5	6.1	6.1
Vanadium	597/597	100	15	150	58	na	nv	nv
Zinc	849/849	100	16	9,700	190	na	410	960
TBT as ion	142/158	90	0.00028 J	3.0	0.09	0.001 – 0.0053	nv	nv

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] RLs are based only on non-detect samples.

CSL – cleanup screening level

RL – reporting limit

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[94] A metal was included if detected concentrations exceeded SMS criteria in at least on surface sediment sample in the LDW.

dw – dry weight
J – estimated concentration
na – not applicable
nv – no value
SMS – Washington State Sediment Management Standards
SQS – sediment quality standards
TBT – tributyltin

Maps 4-50 through 4-59 summarize the concentrations of these metals and TBT in surface sediment samples collected from locations throughout the LDW. Concentrations of metals are shown as individual locations. These data are most relevant on a point-by-point basis, consistent with SMS, because metals other than arsenic are risk drivers for the benthic invertebrate community only. Therefore, only numerical percentiles are discussed. Most of the samples with concentrations that exceeded the 95th percentile[95] for each of the nine metals discussed in this section were located either downstream of RM 2.0 or between RM 3.5 and RM 3.7 on the east side of the navigation channel (within the Boeing Plant 2/Jorgensen Forge EAA). There were several areas of the LDW where the highest concentrations (i.e., greater than the 95th percentile) of various metals were co-located. Most notable of these areas was the Boeing Plant 2/Jorgensen Forge EAA. The maximum concentrations for each of the nine metals (as presented in Table 4-69) were detected in samples collected between RM 3.5 and RM 3.6 within the Boeing Plant 2/Jorgensen Forge EAA.

The Duwamish/Diagonal EAA and its vicinity also had multiple locations with high concentrations of all metals (but most notably cadmium, mercury, and silver). Other metals also had high concentrations at a few locations in this area.

Other areas with concentrations greater than the 95th percentile for multiple metals (Maps 4-50 through 4-58) included:

- One area near RM 0.1 on the eastern shoreline (LDW-SS6, DR001, and LDW-SS305, as identified on Map 4-4a) (cadmium, copper, lead, silver, and zinc)
- One sampling location near RM 1.0 in the navigation channel (LDW-SS37, as identified on Map 4-4c) (cadmium, chromium, mercury, and silver)
- Slip 1 (LDW-SS30, LDW-SS31, LDW-SS32, B3b, and DR018, as identified on Map 4-4c) (cadmium, copper, lead, mercury, nickel, zinc, and vanadium)
- RM 1.3 to RM 1.5 on the west side of the navigation channel (copper, zinc, and vanadium at multiple locations, including LDW-SS48 [see Map 4-4c], which also had concentrations of cadmium, chromium, lead, mercury, and nickel greater than the 95th percentile)

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[95] The SQS for each metal and the SL for nickel were greater than or equal to the corresponding 95th percentile concentrations.

- RM 2.2 on the west side of the navigation channel (LDW-SS84, DR157, and DR139, as identified on Map 4-4e) (cadmium, chromium, lead, mercury, silver, and zinc)
- Slip 4 (R14, DR177, DR178, and DR179, as identified on Map 4-4f) (cadmium, lead, mercury, silver, and vanadium)

Spatial patterns for mercury were different than those for other metals. Additional locations with mercury concentrations greater than the 95th percentile include RM 0.0 to RM 0.1 near Harbor Island, RM 0.3 to RM 0.4 in the channel and to the west of the channel, RM 0.5 to RM 0.6 along the western shoreline, and between RM 0.9 and RM 1.1 in the channel and to the east of the channel. In addition, spatial patterns for TBT were also different than those for the nine metals. The maximum concentration of 3.0 mg/kg dw was detected at sampling location LDW-SS46, located at RM 1.3 on the west side of the LDW (Map 4-59). Additional sampling locations with concentrations of TBT greater than the 95th percentile were in Slip 1 and between RM 0.0 and RM 0.5.

SMS chemical criteria are available for cadmium, chromium, copper, lead, mercury, silver, and zinc. Each of these metals had a relatively small number of SQS or CSL exceedances throughout the LDW compared with the total number of locations sampled; zinc had the highest number of exceedances, with 26 locations (3.1%) exceeding the SQS and 16 locations (1.9%) exceeding the CSL (Table 4-16). The Boeing Plant 2/Jorgensen Forge EAA had the highest number of SQS or CSL exceedances of multiple metals; these exceedances were located primarily between RM 3.5 and RM 3.7. Other areas with SQS or CSL exceedances of multiple metals included the Duwamish/Diagonal EAA, RM 0.5 to RM 0.6 on the west side, Slip 1, RM 1.3 to RM 1.5 on the west side, and EAA 2 at RM 2.2 (Maps 4-50 through 4-58). Outside of these areas, there were three locations (RM 0.1, RM 3.9, and RM 4.8) with lead exceedances (Map 4-53), two locations (RM 1.6 and RM 2.5) with zinc exceedances (Map 4-58), and multiple locations with mercury exceedances (Map 4-54).

4.2.8.2 Subsurface sediment

Subsurface sediment samples collected from 205 cores (471 samples) prior to dredging were analyzed for at least one metal or TBT. Subsurface sediment cores were sampled at various intervals during the different sampling events. To calculate summary statistics, data were categorized according to specified subsurface intervals, as described for total PCBs in Section 4.2.3.2. Table 4-70 presents a summary of subsurface metals and TBT data for samples collected from any interval, and Table 4-71 presents a summary of data for samples collected from specific 1- and 2-ft intervals within the top 10 ft of sediment.

Table 4-70. Summary of data for metals and TBT from any subsurface sediment core interval

	DETECTION FREQUENCY		CONCENTRATION (mg/kg dw)[a]		
CHEMICAL	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLs[b]
Cadmium	257/388	66	0.088	20.4	0.18 – 0.9
Chromium	397/397	100	6.67	386	na
Copper	397/397	100	7.1	2,940	na
Lead	361/394	92	2.8	3,520 J	2 – 3.6
Mercury	341/436	78	0.020	10	0.020 – 0.100
Nickel	317/317	100	4.8	226	na
Silver	166/372	45	0.050	7.5	0.30 – 1
Vanadium	251/251	100	18	223	na
Zinc	396/396	100	16.2 J	4,720	na
Tributyltin as ion	58/95	61	0.00055 J	6.2	0.0010 – 0.0054

Note: Data included in the subsurface dataset are described in Section 4.1.2.2; subsurface sediment samples collected from areas that were subsequently dredged are not summarized in this table because they do not represent existing conditions.

[a] Mean concentrations were not calculated because sampling was generally focused in contaminated areas, resulting in a non-random sampling pattern, which is not likely representative of the entire LDW.

[b] RLs are based only on non-detect samples.

dw – dry weight
J – estimate concentration
LDW – Lower Duwamish Waterway
na – not applicable
RL – reporting limit
TBT – tributyltin

Table 4-71. Summary of subsurface sediment data for metals and TBT in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment

Chemical	Sampling Interval (ft)[a]	Detection Frequency		Concentration (mg/kg dw)[b]		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[c]
Cadmium – 1-ft intervals	0 to 1	52/78	67	0.3	4.5	0.2 – 0.9
	1 to 2	51/80	64	0.3	7.6	0.2 – 0.5
	2 to 3	11/19	58	0.3	18.7	0.20 – 0.38
	3 to 4	1/9	11	0.8	0.8	0.2 – 0.36
	4 to 5	3/9	33	0.50	3.8 J	0.20 – 0.30
	6 to 7	0/5	0	nd	nd	0.20 – 0.30
	8 to 9	1/4	25	0.70	0.70	0.20
Cadmium – 2-ft intervals	0 to 2	72/97	74	0.22	18	0.20 – 0.5
	2 to 4	47/73	64	0.3	15	0.2 – 0.8
	4 to 6	6/6	100	0.7	4.0	na
	6 to 8	3/4	75	1.4	20.4	0.3
	8 to 10	3/3	100	0.12	1.9	0.30
Chromium – 1-ft intervals	0 to 1	86/86	100	6.67	143 J	na
	1 to 2	80/80	100	9.96	135	na
	2 to 3	20/20	100	10.9	300	na
	3 to 4	9/9	100	8.9	24.2	na
	4 to 5	9/9	100	9.9	140 J	na
	6 to 7	5/5	100	10	19.1	na
	8 to 9	4/4	100	8.3	26.5	na
Chromium – 2-ft intervals	0 to 2	97/97	100	11.4 J	210 J	na
	2 to 4	73/73	100	8.1	386	na
	4 to 6	6/6	100	22	92	na
	6 to 8	4/4	100	13.0	160	na
	8 to 10	3/3	100	11.20	54.7	na
Copper – 1-ft intervals	0 to 1	86/86	100	10.9	327	na
	1 to 2	80/80	100	7.1	339	na
	2 to 3	20/20	100	9.20	599	na
	3 to 4	9/9	100	9.00	37.9	na
	4 to 5	9/9	100	7.4	67.3	na
	6 to 7	5/5	100	9.0	26.8	na
	8 to 9	4/4	100	7.5	31.9	na

Table 4-71, cont. Summary of subsurface sediment data for metals and TBT in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment

Chemical	Sampling Interval (ft)[a]	Detection Frequency		Concentration (mg/kg dw)[b]		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[c]
Copper – 2-ft intervals	0 to 2	97/97	100	10.7	800	na
	2 to 4	73/73	100	7.6	2,940	na
	4 to 6	6/6	100	29.7	663	na
	6 to 8	4/4	100	23.2	1,950	na
	8 to 10	3/3	100	15.00	89.4	na
Lead – 1-ft intervals	0 to 1	84/84	100	4.9	639	na
	1 to 2	73/80	91	2.8	514	2.0 – 3
	2 to 3	18/19	95	3	356	2.0
	3 to 4	5/9	56	3	37 J	2 – 3.6
	4 to 5	8/9	89	3.0	64	2.0
	6 to 7	2/5	40	3.0	11	2.0
	8 to 9	1/4	25	43	43	2.0
Lead – 2-ft intervals	0 to 2	95/97	98	3	772	2.5 – 3
	2 to 4	65/73	89	3	3,520 J	2 – 3
	4 to 6	6/6	100	18.0	1,210	na
	6 to 8	4/4	100	42	1,350	na
	8 to 10	3/3	100	3.00	89	na
Mercury – 1-ft intervals	0 to 1	81/84	96	0.05	0.71	0.05 – 0.100
	1 to 2	54/66	82	0.06	0.6	0.040 – 0.0600
	2 to 3	13/19	68	0.070	0.64	0.030 – 0.060
	3 to 4	2/9	22	0.07	0.17	0.020 – 0.06
	4 to 5	3/10	30	0.060	0.24 J	0.040 – 0.060
	6 to 7	2/5	40	0.050	0.16	0.040 – 0.050
	8 to 9	1/3	33	0.19	0.19	0.050
Mercury – 2-ft intervals	0 to 2	102/107	95	0.05	10	0.04 – 0.07
	2 to 4	68/84	81	0.07	1.4	0.020 – 0.06
	4 to 6	9/15	60	0.080	0.98	0.050 – 0.060
	6 to 8	6/15	40	0.075	4.34	0.040 – 0.060
	8 to 10	4/12	25	0.25	0.89	0.050 – 0.080
Nickel – 1-ft intervals	0 to 1	65/65	100	4.8	36	na
	1 to 2	53/53	100	6.92	51.1	na
	2 to 3	10/10	100	9	32	na
	3 to 4	9/9	100	6 J	25	na
	8 to 9	1/1	100	5	5	na

Table 4-71, cont. Summary of subsurface sediment data for metals and TBT in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment

CHEMICAL	SAMPLING INTERVAL (ft)[a]	DETECTION FREQUENCY		CONCENTRATION (mg/kg dw)[b]		
		RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLs[c]
Nickel – 2-ft intervals	0 to 2	76/76	100	7 J	37	na
	2 to 4	70/70	100	6	226	na
	4 to 6	5/5	100	13	34	na
	6 to 8	4/4	100	10	69	na
	8 to 10	3/3	100	12.00	24	na
Silver – 1-ft intervals	0 to 1	15/62	24	0.500	3.0	0.4 – 1
	1 to 2	23/80	29	0.5	2.6	0.3 – 1
	2 to 3	8/19	42	0.060	7.3	0.40 – 0.5
	3 to 4	1/9	11	0.61	0.61	0.3 – 0.5
	4 to 5	1/9	11	1.2	1.2	0.30 – 0.40
	6 to 7	0/5	0	nd	nd	0.30 – 0.40
	8 to 9	0/4	0	nd	nd	0.30 – 0.40
Silver – 2-ft intervals	0 to 2	52/97	54	0.050	3.3	0.4 – 1
	2 to 4	34/73	47	0.080	5	0.3 – 1
	4 to 6	4/6	67	1.4	4.3	0.400 – 1
	6 to 8	3/4	75	2	3	0.4
	8 to 10	2/3	67	0.5	2.3	0.40
Vanadium – 1-ft intervals	0 to 1	64/64	100	18	85	na
	1 to 2	37/37	100	39.6	84.3	na
	2 to 3	9/9	100	47.0	67	na
	3 to 4	8/8	100	36.1	71.5	na
	8 to 9	1/1	100	37.5	37.5	na
Vanadium – 2-ft intervals	0 to 2	73/73	100	37	87	na
	2 to 4	68/68	100	26	223	na
	4 to 6	5/5	100	39.8	95.1	na
	6 to 8	4/4	100	41.0	112	na
	8 to 10	2/2	100	63.9	75.4	na
Zinc – 1-ft intervals	0 to 1	86/86	100	18.6	1,260	na
	1 to 2	80/80	100	20.2	2,050	na
	2 to 3	19/19	100	22.3	1,770	na
	3 to 4	9/9	100	18.5	69	na
	4 to 5	9/9	100	20.8	324 J	na
	6 to 7	5/5	100	22.0	47.0	na
	8 to 9	4/4	100	17.6	84.7	na

Table 4-71, cont. Summary of subsurface sediment data for metals and TBT in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment

CHEMICAL	SAMPLING INTERVAL (ft)[a]	DETECTION FREQUENCY		CONCENTRATION (mg/kg dw)[b]		
		RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLs[c]
Zinc –2-ft intervals	0 to 2	97/97	100	21	1,660	na
	2 to 4	73/73	100	16.2 J	4,720	na
	4 to 6	6/6	100	69.0	1,430	na
	6 to 8	4/4	100	88.4	4,550	na
	8 to 10	3/3	100	21.00	186	na
Tributyltin as ion – 1-ft intervals	0 to 1	11/11	100	0.0055	0.22	na
	1 to 2	6/14	43	0.021	0.35	0.0036 – 0.0054
	2 to 3	0/1	0	nd	nd	0.0010
	3 to 4	0/1	0	nd	nd	0.0036
	8 to 9	0/1	0	nd	nd	0.0036
Tributyltin as ion – 2-ft intervals	0 to 2	22/27	81	0.0030	2.5	0.0010 – 0.0049
	2 to 4	16/26	62	0.010	2.1	0.0010 – 0.0038
	4 to 6	3/3	100	0.014	1.0	na
	6 to 8	2/2	100	3.4	6.2	na
	8 to 10	0/1	0	nd	nd	0.0052569

Note: Data included in the subsurface dataset are described in Section 4.1.2.2; subsurface sediment samples collected from areas that were subsequently dredged are not summarized in this table because they do not represent existing conditions.

[a] For the calculation of summary statistics in this table, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to the 1- or 2-ft sampling category that best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were averaged to obtain a calculated concentration for the corresponding 2-ft interval. Some of the subsurface data were collected from intervals other than 1 or 2 ft; therefore, the detection frequency ratio for "any interval" in Table 4-70 may reflect data for intervals not included in these categories.

[b] Mean concentrations were not calculated because sampling was generally focused in contaminated areas, resulting in a non-random sampling pattern, which is not likely representative of the entire LDW.

[c] RLs based only on non-detect samples.

dw – dry weight

J – estimated concentration

na – not applicable

nd – not detected

RL – reporting limit

Over 90% of the subsurface sediment samples summarized in Table 4-71 were collected from the top 4 ft of sediment. There were relatively few samples analyzed from depths greater than 4 ft below mudline; most of those samples were collected and analyzed as part of the RI at locations with SMS exceedances in upper intervals or during other investigations targeting specific locations.

The highest copper and zinc concentrations were detected in a subsurface sediment sample collected from the head of Slip 3 and the highest cadmium concentration was in a subsurface sediment sample collected from the head of Slip 1. The highest concentrations of lead, chromium, and silver in any subsurface sediment samples (Table 4-70) were detected in the 0-to-3-ft or 0-to-1-ft intervals within or just outside of the Duwamish/Diagonal EAA. In this area, high metal concentrations were detected in samples collected from both uppermost core intervals as well as deeper intervals within the cores. High metal concentrations were also detected in subsurface samples from both uppermost intervals and at depth in the Boeing Plant 2/Jorgensen Forge EAA.

Other notable areas with subsurface metal concentrations greater than the 95th percentile for multiple metals include:

- RM 0.1 on the east side (LDW-SC2, as identified on Map 4-7a) (cadmium and silver, with peak concentrations in the 2-to-4-ft interval and lead and zinc, with peak concentrations in the 4-to-6-ft interval)
- RM 0.5 on the western shoreline (LDW-SC11, as identified on Map 4-7a) (cadmium, copper, lead, mercury, and zinc, with peak concentrations in the 0-to-1-ft interval)
- Slip 1 (DR021, LDW-SC16 and LDW-SC17, as identified on Map 4-7a) (up to nine metals in each core, with peak concentrations at intervals of 2 to 4 ft, 4 to 6 ft, or 6 to 8 ft)
- RM 1.3 to RM 1.5 on the west side of the navigation channel (DR054, LDW-SC25, LDW-SC26, LDW-SC27, and LDW-SC28, as identified on Map 4-7b) (up to eight metals in each core, with peak concentrations primarily at intervals of 2 to 4 ft, 4 to 6 ft, or 6 to 8 ft, but some high concentrations were also in the 0-to-2-ft interval)
- Slip 3 (LDW-SC37, as identified on Map 4-7b) (nine metals, with peak concentrations primarily in the 2-to-4-ft interval)
- RM 3.9 on the east side (LDW-SC52, as identified on Map 4-7d) (lead and mercury, with peak concentrations in the 0-to-1-ft interval; chromium, nickel, and silver, with peak concentrations in the 1-to-2-ft interval; and cadmium with the same concentration in both the 0-to-1-ft and 1-to-2-ft intervals)

TBT concentrations in subsurface sediment greater than the 95th percentile were located primarily near RM 1.3 and RM 1.4 on the west side (Map 4-60). The two highest TBT concentrations (6.2 and 3.4 mg/kg dw) were in the 6-to-8-ft interval in cores LDW-SC26 and LDW-SC28, respectively, collected in this area.

4.2.8.3 Tissue

Table 4-72 presents the detection frequencies and ranges of concentrations of seven metals and TBT that were identified as COCs in specific tissue types.[96] Appendix E, Section E.6, presents a summary of data for other metals and inorganic substances that were analyzed in tissue but are not discussed in this section.

The tissue types with the highest mean concentrations of each metal (including TBT) were invertebrates (i.e., clams, crabs, benthic invertebrates, and amphipods). In general, fish fillet samples had the lowest mean concentrations of TBT and metals, except for mercury. Mean concentrations in whole-body fish samples were higher than those in fish fillets for TBT and all metals, except mercury. In crab hepatopancreas, mean concentrations of cadmium, copper, lead, and TBT were higher than in crab edible meat.

TBT concentrations were higher in soft-shell clams relative to other tissue types. TBT concentrations in the 14 clam samples ranged from 0.15 to 0.66 mg/kg ww; whereas the mean concentrations in other tissue types ranged from 0.0033 to 0.029 mg/kg ww.

As part of the ERA, regression analyses were conducted using co-located sediment and benthic invertebrate tissue data for metals and TBT (see Appendix A, Attachment 11). TBT was the only analyte with a significant relationship ($R^2 = 0.59$, $p < 0.05$). However, there is uncertainty in this relationship because of one sediment-tissue pair that was highly influential (Appendix A, Attachment 11).

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[96] A metal is discussed in Section 4.2.7.3 for tissue if it was identified as a COC in the HHRA (Appendix B) based on seafood consumption or if the LOAEL-based HQ was ≥ 1.0 for any fish or wildlife receptor in the ERA (Appendix A).

Table 4-72. Summary of metals and TBT data in tissue composite samples

Tissue Type[a]	Unit	Detection Frequency		Concentration			
		Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	RL or Range of RLs[c]
Cadmium							
Crab, edible meat	mg/kg ww	21/21	100	0.0055	0.0444	0.023	na
Crab, hepatopancreas	mg/kg ww	8/8	100	0.11	0.8530	0.48	na
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.04 M	0.2951 M	0.16	na
Fish, fillet (with or without skin)	mg/kg ww	1/13	8	0.0013 J	0.0013 J	nc	0.0042 – 0.0079
Fish, whole body	mg/kg ww	75/75	100	0.002 J	0.0240	0.01	na
Amphipod, whole body	mg/kg ww	4/4	100	0.017	0.15	0.058	na
Benthic invertebrates, whole body	mg/kg ww	20/20	100	0.0175 J	0.202 J	0.060	na
Mussels, wild (edible meat)	mg/kg ww	22/22	100	0.19	0.84	0.49	na
Mussels, transplanted (edible meat)	mg/kg ww	35/35	100	0.231	0.781	0.379	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.064	0.148	0.10	na
Chromium							
Crab, edible meat	mg/kg ww	2/21	10	0.13	0.16	0.05	0.07 – 0.11
Crab, hepatopancreas	mg/kg ww	3/8	38	0.04 J	0.16	0.05	0.01 – 0.08
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.07 JM	0.140 M	0.04	0.03 – 0.05
Fish, fillet (with or without skin)	mg/kg ww	2/13	15	0.054	0.062	nc	0.049 – 0.12
Fish, whole body	mg/kg ww	53/75	71	0.06 J	3.74	0.2	0.10 – 0.14
Amphipod, whole body	mg/kg ww	4/4	100	0.45 J	0.56 J	0.52	na
Benthic invertebrates, whole body	mg/kg ww	20/20	100	0.08	3.90	1.0	na
Mussels, wild (edible meat)	mg/kg ww	21/22	95	0.10	0.35	0.16	0.050
Mussels, transplanted (edible meat)	mg/kg ww	33/35	94	0.059	0.305	0.1	0.05
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.36	1.32	0.67	na
Copper							
Crab, edible meat	mg/kg ww	21/21	100	4.430	16	7.5	na
Crab, hepatopancreas	mg/kg ww	8/8	100	12.4	49.9	28	na
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	6.9 M	24 M	14	na

Table 4-72, cont. Summary of metals and TBT data in tissue composite samples

		DETECTION FREQUENCY		CONCENTRATION			
TISSUE TYPE[a]	UNIT	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	CALCULATED MEAN[b]	RL OR RANGE OF RLS[c]
Fish, fillet (with or without skin)	mg/kg ww	19/19	100	0.18	1.390	0.67	na
Fish, whole body	mg/kg ww	75/75	100	0.494	3.470	1.4	na
Amphipod, whole body	mg/kg ww	4/4	100	9.8	30	19	na
Benthic invertebrates, whole body	mg/kg ww	20/20	100	1.940 J	21.8	8.52	na
Mussels, wild (edible meat)	mg/kg ww	22/22	100	0.58	1.7 J	1.2	na
Mussels, transplanted (edible meat)	mg/kg ww	35/35	100	0.513	2.08	1.13	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	3.50	7.30	5.34	na
Lead							
Crab, edible meat	mg/kg ww	21/21	100	0.012	0.24 J	0.045	na
Crab, hepatopancreas	mg/kg ww	8/8	100	0.037	0.2690	0.11	na
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.020 M	0.22 JM	0.067	na
Fish, fillet (with or without skin)	mg/kg ww	10/19	53	0.006	0.137	0.04	0.020 – 0.030
Fish, whole body	mg/kg ww	75/75	100	0.012	0.946	0.17	na
Amphipod, whole body	mg/kg ww	4/4	100	0.95	7.4	3.7	na
Benthic invertebrates, whole body	mg/kg ww	20/20	100	0.1430	14.60	1.77	na
Mussels, wild (edible meat)	mg/kg ww	22/22	100	0.13	0.72	0.41	na
Mussels, transplanted (edible meat)	mg/kg ww	35/35	100	0.043	0.288	0.13	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.368	6.370	1.96	na
Mercury							
Crab, edible meat	mg/kg ww	25/25	100	0.023	0.11	0.057	na
Crab, hepatopancreas	mg/kg ww	8/8	100	0.020	0.067	0.031	na
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.022 M	0.097 M	0.046	na
Fish, fillet (with or without skin)	mg/kg ww	27/27	100	0.013	0.083	0.040	na
Fish, whole body	mg/kg ww	81/81	100	0.005	0.088	0.03	na
Amphipod, whole body	mg/kg ww	4/4	100	0.0067	0.017	0.011	na
Benthic invertebrates, whole body	mg/kg ww	18/20	90	0.002 J	0.044	0.01	0.004 – 0.009
Mussels, wild (edible meat)	mg/kg ww	21/21	100	0.0088	0.023	0.013	na
Mussels, transplanted (edible meat)	mg/kg ww	35/35	100	0.0051	0.0088	0.0066	na

Table 4-72, cont. Summary of metals and TBT data in tissue composite samples

Tissue Type[a]	Unit	Detection Frequency		Concentration			
		Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	RL or Range of RLs[c]
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.009	0.022	0.02	na
Vanadium							
Crab, edible meat	mg/kg ww	0/19	0	nd	nd	nc	0.14 – 0.21
Crab, hepatopancreas	mg/kg ww	4/7	57	0.11 J	0.2 J	0.1	0.13 – 0.21
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.13 M	0.2 JM	0.1	0.08 – 0.11
Fish, fillet (with or without skin)	mg/kg ww	0/10	0	nd	nd	nc	0.2 – 0.25
Fish, whole body	mg/kg ww	56/72	78	0.1 J	1.23	0.3	0.2 – 0.25
Benthic Invertebrates,	mg/kg ww	20/20	100	0.26	3.04	1.2	na
Mussels, wild (edible meat)	mg/kg ww	8/8	100	0.058	0.26	0.15	na
Mussels, transplanted (edible meat)	mg/kg ww	10/10	100	0.119	0.281	0.195	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.68	2.65	1.3	na
Zinc							
Crab, edible meat	mg/kg ww	21/21	100	26.1	39.3	34	na
Crab, hepatopancreas	mg/kg ww	8/8	100	14.8	33.6	23	na
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	24.6 M	37.3 M	31	na
Fish, fillet (with or without skin)	mg/kg ww	13/13	100	3.8	9.33	7.0	na
Fish, whole body	mg/kg ww	75/75	100	9.95	28.0	16	na
Amphipod, whole body	mg/kg ww	4/4	100	7.9	26	14	na
Benthic invertebrates, whole body	mg/kg ww	20/20	100	8.22	43.6	20.9	na
Mussels, wild (edible meat)	mg/kg ww	22/22	100	17	44	30	na
Mussels, transplanted (edible meat)	mg/kg ww	35/35	100	4.78	20.6	9.71	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	16.1	32.3	23.7	na
Tributyltin as ion							
Crab, edible meat	mg/kg ww	9/25	36	0.00081 J	0.082	0.0062	0.0015 – 0.0020
Crab, hepatopancreas	mg/kg ww	6/8	75	0.00050 J	0.059	0.019	0.0015
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.00120 JM	0.075 M	0.0099	0.0008
Fish, fillet (with or without skin)	mg/kg ww	14/21	67	0.0012 J	0.016 J	0.0033	0.00074 – 0.0020
Fish, whole body	mg/kg ww	78/92	85	0.0018 J	0.18	0.027	0.0015 – 0.0021

Table 4-72, cont. Summary of metals and TBT data in tissue composite samples

Tissue Type[a]	Unit	Detection Frequency		Concentration			
		Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	RL or Range of RLs[c]
Amphipod, whole body	mg/kg ww	4/4	100	0.018	0.036	0.029	na
Benthic invertebrates, whole body	mg/kg ww	19/20	95	0.0038 J	0.092	0.028	0.0050
Mussels, wild (edible meat)	mg/kg ww	22/22	100	0.012	0.037	0.023	na
Mussels, transplanted (edible meat)	mg/kg ww	32/32	100	0.00935	0.0369	0.0196	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.15	0.66	0.32	na

Note: A metal is included in this table if it was detected in any tissue sample and was identified as a COC in the HHRA (Appendix B) based on seafood consumption or if it was a COC for any fish or wildlife receptor in the ERA (Appendix A).

[a] Crab tissue includes a combination of Dungeness, slender, and red rock crabs. Fish tissue includes English sole, Pacific staghorn sculpin, pile perch, shiner surfperch, starry flounder, and striped perch, as available.

[b] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results. Mean concentrations were not calculated if the detection frequency was ≤ 25%.

[c] RLs are based only on non-detect samples.

[d] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

COC – chemical of concern
ERA – ecological risk assessment
HHRA – human health risk assessment
J – estimated concentration
M – calculated whole-body concentration
na – not applicable
nc – not calculated
nd – not detected
RL – reporting limit
TBT – tributyltin
ww – wet weight

4.2.8.4 Surface water

Table 4-73 presents a summary of metal concentrations in LDW surface water samples collected as part of the King County WQA (Map 4-11a). Chromium, lead, vanadium, and zinc were detected in every surface water sample. Silver was not detected in any sample. TBT was not analyzed in surface water samples. Mean concentrations of chromium, copper, lead, nickel, vanadium, and zinc in unfiltered samples and copper, lead, and zinc in filtered samples were higher in samples from RM 4.9 than in samples from RM 1.1 and RM 1.9. Concentrations of metals in surface water samples are compared with WQC in Section 4.2.2 (Table 4-20).

Table 4-73. Summary of surface water data for metals from three locations in the LDW

LOCATION ID[a]	DETECTION FREQUENCY		CONCENTRATION (µg/L)		
	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLs[b]
Cadmium					
Filtered					
BRN (RM 1.1)	72/72	100	0.011	0.0755	na
SWM (RM 1.9)	72/72	100	0.0079	0.0795	na
NFK (RM 4.9)	12/22	55	0.0072	0.021	0.0067 – 0.0073
Unfiltered					
BRN (RM 1.1)	173/173	100	0.0088	0.0780	na
SWM (RM 1.9)	171/173	99	0.0076	0.391	0.0067 – 0.0068
NFK (RM 4.9)	49/56	88	0.0071	0.032	0.0068 – 0.0073
Chromium					
Filtered					
BRN (RM 1.1)	68/68	100	0.11 J	0.576 J	na
SWM (RM 1.9)	56/56	100	0.10 J	0.453 J	na
NFK (RM 4.9)	24/24	100	0.14 J	0.423 J	na
Unfiltered					
BRN (RM 1.1)	165/165	100	0.240 J	1.74 J	na
SWM (RM 1.9)	157/157	100	0.277 J	2.32 J	na
NFK (RM 4.9)	58/58	100	0.256 J	2.37 J	na
Copper					
Filtered					
BRN (RM 1.1)	65/65	100	0.384 J	1.54 J	na
SWM (RM 1.9)	64/66	97	0.370 J	1.89 J	0.354 – 0.368
NFK (RM 4.9)	23/24	96	0.496 J	1.34 J	0.628
Unfiltered					
BRN (RM 1.1)	166/166	100	0.536 J	5.83 J	na
SWM (RM 1.9)	166/167	99	0.474 J	4.03 J	0.643

LOCATION ID[a]	DETECTION FREQUENCY RATIO	%	CONCENTRATION (µg/L) MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLs[b]
NFK (RM 4.9)	58/58	100	0.728 J	4.24 J	na
Lead					
Filtered					
BRN (RM 1.1)	70/70	100	0.010 J	0.553 J	na
SWM (RM 1.9)	66/66	100	0.0077 J	0.198 J	na
NFK (RM 4.9)	24/24	100	0.0611 J	0.343 J	na
Unfiltered					
BRN (RM 1.1)	171/171	100	0.0459 J	1.45 J	na
SWM (RM 1.9)	167/167	100	0.0570 J	1.57 J	na
NFK (RM 4.9)	58/58	100	0.143 J	2.81 J	na
Mercury					
Filtered[c]					
BRN (RM 1.1)	8/9	89	0.00013	0.00051	0.0001
NFK (RM 4.9)	6/6	100	0.00046	0.00071	na
Unfiltered					
BRN (RM 1.1)	9/15	60	0.00056	0.00336	0.20
SWM (RM 1.9)	0/6	0	nd	nd	0.20
NFK (RM 4.9)	6/8	75	0.00104	0.00689	0.20
Nickel					
Filtered					
BRN (RM 1.1)	61/65	94	0.274 J	0.910 J	0.154 – 0.387
SWM (RM 1.9)	54/60	90	0.259 J	1.50 J	0.118 – 0.306
NFK (RM 4.9)	21/21	100	0.190 J	0.472 J	na
Unfiltered					
BRN (RM 1.1)	145/147	99	0.394 J	2.11 J	0.348 – 0.386
SWM (RM 1.9)	137/143	96	0.328 J	2.88 J	0.291 – 0.575
NFK (RM 4.9)	55/55	100	0.434 J	2.91 J	na
Silver					
Filtered					
BRN (RM 1.1)	0/72	0	nd	nd	0.11 – 0.13
SWM (RM 1.9)	0/72	0	nd	nd	0.097 – 0.13
NFK (RM 4.9)	0/24	0	nd	nd	0.10 – 0.13
Unfiltered					
BRN (RM 1.1)	0/173	0	nd	nd	0.10 – 0.13
SWM (RM 1.9)	0/173	0	nd	nd	0.10 – 0.13
NFK (RM 4.9)	0/58	0	nd	nd	0.11 – 0.13

Location ID[a]	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
Vanadium					
Filtered					
BRN (RM 1.1)	50/50	100	0.193	1.57	na
SWM (RM 1.9)	42/42	100	0.370	1.56	na
NFK (RM 4.9)	18/18	100	0.141	0.400	na
Unfiltered					
BRN (RM 1.1)	133/133	100	0.267	2.96	na
SWM (RM 1.9)	125/125	100	0.220	3.99	na
NFK (RM 4.9)	46/46	100	0.315	3.57	na
Zinc					
Filtered					
BRN (RM 1.1)	72/72	100	0.750 J	5.39 J	na
SWM (RM 1.9)	72/72	100	0.825 J	4.09 J	na
NFK (RM 4.9)	22/22	100	0.881 J	5.24 J	na
Unfiltered					
BRN (RM 1.1)	173/173	100	0.70 J	8.34 J	na
SWM (RM 1.9)	173/173	100	1.08 J	6.62 J	na
NFK (RM 4.9)	56/56	100	0.979 J	9.04 J	na

[a] BRN, SWM, and NFK are located in the LDW near the Brandon, SW Michigan, and Norfolk CSOs, respectively.

[b] RLs are based only on non-detect samples.

[c] Mercury data are not available for filtered samples from the SWM (RM 1.9) location.

J – estimated concentration
na – not applicable
nc – not calculated
nd – not detected
RM – river mile

4.2.8.5 Seep water

Seep water samples were collected and analyzed for metals during four sampling events (Table 4-74; Map 4-11a). At Boeing Plant 2, 10 filtered seep water samples and 17 unfiltered samples were analyzed for metals. During the Rhône-Poulenc sampling event, unfiltered seep water samples from seven locations were analyzed for metals, with only detected concentrations reported. At T-117, three filtered and three unfiltered seep water samples were analyzed for metals. As part of the LDW-wide RI sampling event, 16 filtered and 13 unfiltered seep water samples were analyzed for metals. No seep water samples were analyzed for TBT during any of the four sampling events. Detected concentrations of metals in seep water samples are shown on Maps 4-11b through 4-11e.

Table 4-74. Summary of seep water data for metals

Chemical	Detection Frequency		Concentration (µg/L)			Location of Maximum Detect[b]
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]	
Cadmium						
Boeing Plant 2						
Cadmium (filtered)	0/10	0	nd	nd	2 – 10	na
Cadmium (unfiltered)	0/17	0	nd	nd	2 – 10	na
Rhône-Poulenc						
Cadmium (unfiltered)	2/7	29	2.1	11.6	nr	02-SP
T-117						
Cadmium (filtered)	0/3	0	nd	nd	2.00	na
Cadmium (unfiltered)	0/3	0	nd	nd	2.0	na
LDW RI						
Cadmium (filtered)	16/16	100	0.009	0.508	na	SP-82
Cadmium (unfiltered)	13/13	100	0.022	0.710	na	SP-54
Chromium						
Boeing Plant 2						
Chromium (filtered)	0/10	0	nd	nd	5 – 20	na
Chromium (unfiltered)	6/17	35	6	49	5 – 20	SE-SWY03
Rhône-Poulenc						
Chromium (unfiltered)	1/7	14	202	202	nr	02-SP
T-117						
Chromium (filtered)	2/3	67	8.00 J	9.00 J	5.00	SEEP_2
Chromium (unfiltered)	2/3	67	6.00	7.00	5.00	SEEP_3
LDW RI						
Chromium (filtered)	0/16	0	nd	nd	1.51 – 9.74	na
Chromium (unfiltered)	1/13	8	74.9	74.9	0.07 – 11.4	SP-54
Copper						
Boeing Plant 2						
Copper (filtered)	1/10	10	8	8	2 – 10	SE-94105
Copper (unfiltered)	11/17	65	2	60	2 – 10	SE-SWY03
Rhône-Poulenc						
Copper (unfiltered)	1/7	14	203	203	nr	02-SP
T-117						
Copper (filtered)	3/3	100	4 J	5.00 J	na	SEEP_2, SEEP_1
Copper (unfiltered)	3/3	100	2.50	4.00	na	SEEP_3
LDW RI						
Copper (filtered)	7/16	44	8.16 J	22.8	3.28 – 7.77	SP-80
Copper (unfiltered)	10/13	77	8.06 J	50.9	4.75 – 6.47	SP-76

Table 4-74, cont. Summary of seep water data for metals

Chemical	Detection Frequency		Concentration (µg/L)			Location of Maximum Detect[b]
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]	
Lead						
Boeing Plant 2						
Lead (filtered)	0/10	0	nd	nd	1 – 20	na
Lead (unfiltered)	14/17	82	1	104	20	SE-SWY02
Rhône-Poulenc						
Lead (unfiltered)	1/7	14	44.1	44.1	nr	02-SP
T-117						
Lead (filtered)	0/3	0	nd	nd	20	na
Lead (unfiltered)	0/3	0	nd	nd	20	na
LDW RI						
Lead (filtered)	16/16	100	0.036	3	na	SP-76
Lead (unfiltered)	13/13	100	0.080	296	na	SP-54
Mercury						
Boeing Plant 2						
Mercury (filtered)	0/10	0	nd	nd	0.1	na
Mercury (unfiltered)	3/17	18	0.2	0.2	0.1	SE-31003, SE-SWY01, SE-SWY03
Rhône-Poulenc						
Mercury (unfiltered)	1/7	14	0.65	0.65	nr	05-SP
T-117						
Mercury (unfiltered)	0/3	0	nd	nd	0.10	na
LDW RI						
Mercury (filtered)	16/16	100	0.00062	0.0153	na	SP-76
Mercury (unfiltered)	13/13	100	0.00061	0.582	na	SP-54
Nickel						
Boeing Plant 2						
Nickel (filtered)	0/10	0	nd	nd	10 – 50	na
Nickel (unfiltered)	4/17	24	10	50	10 – 50	SE-SWY04
Rhône-Poulenc						
Nickel (unfiltered)	1/7	14	70.0	70.0	nr	02-SP
LDW RI						
Nickel (filtered)	12/16	75	0.84	5.25	0.040	SP-20
Nickel (unfiltered)	12/13	92	2.80	8.83	0.040	SP-20
Silver						
Boeing Plant 2						
Silver (filtered)	0/10	0	nd	nd	3 – 20	na
Silver (unfiltered)	0/17	0	nd	nd	3 – 20	na

Table 4-74, cont. Summary of seep water data for metals

Chemical	Detection Frequency		Concentration (µg/L)			Location of Maximum Detect[b]
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]	
T-117						
Silver (filtered)	0/3	0	nd	nd	3.00	na
Silver (unfiltered)	0/3	0	nd	nd	3.00	na
LDW RI						
Silver (filtered)	14/16	88	0.012	0.112	0.015	SP-20
Silver (unfiltered)	11/13	85	0.025	0.11	0.015	SP-82
Vanadium						
Boeing Plant 2						
Vanadium (filtered)	2/10	20	2	4	2 – 10	SE-11001
Vanadium (unfiltered)	15/17	88	2	50	2 – 10	SE-31003
Rhône-Poulenc						
Vanadium (unfiltered)	7/7	100	2.8	400	na	02-SP
Zinc						
Boeing Plant 2						
Zinc (filtered)	3/10	30	30	90	4 – 20	SE-SWY04
Zinc (unfiltered)	14/17	82	6	200	20	SE-SWY04
Rhône-Poulenc						
Zinc (unfiltered)	1/7	14	223	223	nr	02-SP
T-117						
Zinc (filtered)	0/3	0	nd	nd	6.00	na
Zinc (unfiltered)	1/3	33	7.00 J	7.00 J	6.00	SEEP_1
LDW RI						
Zinc (filtered)	16/16	100	3.29	161	na	SP-82
Zinc (unfiltered)	13/13	100	3.49	322	na	SP-54

[a] RLs are based only on non-detect samples.

[b] Sampling locations are shown on Map 4-11a.

J – estimated concentration
na – not applicable
nc – not calculated
nd – not detected
nr – not reported; RLs for non-detects not reported for this sampling event
RI – remedial investigation
RL – reporting limit

The unfiltered seep water sample collected from location 02-SP near Slip 6 (Map 4-11e) had the highest concentrations of cadmium, chromium, copper, nickel, and vanadium of any samples collected during the four sampling events. Concentrations of metals in seep water samples are compared with WQC in Section 4.2.2 (Table 4-21).

4.2.8.6 Porewater

Metals were analyzed in 15 unfiltered porewater samples collected during the 2004 Rhône-Poulenc sampling event and in 15 unfiltered porewater samples collected throughout the LDW during the 1998 EPA SI (Table 4-75). In addition, TBT was analyzed in 20 unfiltered samples collected by USACE between RM 1.8 and RM 3.2 in 1999. Concentrations of the seven metals detected in individual porewater samples are shown on Maps 4-12b through 4-12d and are compared with WQC in Section 4.2.2 (Table 4-22). TBT was detected in 95% of the porewater samples at concentrations ranging from 0.010 to 0.15 µg/L, with the exception of one sample collected near RM 2.5 that had a concentration of 0.55 µg/L, which was greater than the DMMP SL of 0.15 µg/L (Maps 4-12b and 4-12c).

Table 4-75. Summary of porewater data for detected metals

Chemical	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Rhône-Poulenc					
Total copper	4/15	27	5.4	264	5
Total mercury	10/15	67	0.0016	0.408	0.2
Total zinc	3/15	20	263	1,560	50
EPA SI					
Total cadmium	4 /15	27	4 J	4 J	5
Total copper	4/15	27	1	5	4
Total lead	13/15	87	0.6 J	4	1
Total silver	6/15	40	0.3 J	0.5 J	1
Total vanadium	15/15	100	3 J	22	na
Total zinc	3/15	20	4 J	6 J	10
PSDDA99					
TBT	19/20	95	0.010 J	0.55	0.020

[a] RLs are based only on non-detect samples.

EPA – US Environmental Protection Agency
J – estimated concentration
na – not applicable
PSDDA – Puget Sound Dredged Disposal Analysis
RL – reporting limit
SI – sediment investigation
TBT – tributyltin

4.2.8.7 Summary of metals and TBT data

Most samples with concentrations greater than the numerical 95th percentile for each metal were located either downstream of RM 2.2 or between RM 3.5 and RM 3.7 (within the Boeing Plant 2/Jorgensen Forge EAA). The Duwamish/Diagonal EAA and the Boeing Plant 2/Jorgensen Forge EAA had many of the highest metals concentrations in surface sediment. Other notable areas with high concentrations of more than one metal in surface sediment include Slip 1, Slip 4, RM 1.3 to RM 1.5 on the west side, and RM 2.2 on the west side. TBT concentrations in surface sediment were highest near RM 1.3 on the west side of the LDW and downstream of RM 1.0.

In tissue, the highest mean concentrations of metals (including TBT) were detected in invertebrate species (i.e., clams, crabs, benthic invertebrates, and amphipods). Mean concentrations of metals (except mercury) and TBT in whole-body fish samples were higher than those in fish fillets. In crab hepatopancreas, mean concentrations of cadmium, copper, lead, and TBT were higher than those in crab edible meat.

In surface water, higher mean concentrations of chromium, copper, lead, nickel, vanadium, and zinc in unfiltered samples and copper, lead, and zinc in filtered samples were detected in samples from RM 4.9 than in samples from RM 1.1 and RM 1.9. In seep water, one unfiltered sample collected near Slip 6 had the highest concentrations of cadmium, chromium, copper, nickel, and vanadium. Thirty unfiltered porewater samples collected from throughout the LDW were analyzed for metals; seven metals were detected in these samples. Twenty porewater samples were collected between RM 1.8 and RM 3.2 and analyzed for TBT. The highest TBT concentration (0.55 µg/L) was detected in a sample collected near RM 2.5. This was the only sample with a TBT concentration that exceeded the DMMP SL of 0.15 µg/L.

4.2.9 SVOCs

This section summarizes the nature and extent of SVOC concentrations in surface sediment, subsurface sediment, tissue, surface water, seep water, and porewater. The nature and extent of cPAH and BEHP concentrations is discussed in Sections 4.2.5 and 4.2.6, respectively.

4.2.9.1 Surface sediment

Table 4-76 presents detection frequencies and a summary of concentrations in surface sediment for selected SVOCs.[97] Table 4-77 presents detection frequencies and a summary of OC-normalized concentrations for SVOCs with SMS criteria expressed in units of mg/kg OC. Surface sediment samples collected from 828 of the 1,365 locations in the RI baseline dataset were analyzed for one or more PAHs, samples from 832 locations were analyzed for one or more phthalates, and samples from 831 locations were analyzed for at least one other SVOC included in Table 4-76.

Table 4-76. Summary of surface sediment data for selected SVOCs

	DETECTION FREQUENCY		CONCENTRATION (µg/kg dw)				SMS CRITERIA (µg/kg dw)	
CHEMICAL	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	CALCULATED MEAN[a]	RL OR RANGE OF RLs[b]	SQS	CSL
PAHs								
2-Methylnaphthalene	139/818	17	1.0 J	3,300	nc	1.0 – 2,000	na[c]	na[c]
Acenaphthene	304/828	37	1.0 J	5,200	70	1.8 – 2,000	na[c]	na[c]
Anthracene	576/828	70	2.0	10,000	100	13 – 2,000	na[c]	na[c]

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[97] An SVOC was included in Section 4.2.8.1 if it exceeded the SQS in at least one surface sediment sample or if it was identified as a COC in the HHRA based on a sediment exposure pathway.

	DETECTION FREQUENCY		CONCENTRATION (µg/kg dw)				SMS CRITERIA (µg/kg dw)	
CHEMICAL	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	CALCULATED MEAN[a]	RL OR RANGE OF RLs[b]	SQS	CSL
Benzo(a)anthracene	748/828	90	7.3 J	8,400	320	6.4 – 200	na[c]	na[c]
Benzo(a)pyrene	747/822	91	6.5	7,900	310	6.4 – 350	na[c]	na[c]
Benzo(g,h,i)perylene	682/823	83	6.1	3,800	200	13 – 2,000	na[c]	na[c]
Total benzofluoranthenes	757/822	92	6.6 J	17,000	740	nc	na[c]	na[c]
Chrysene	773/828	93	12	7,700	500	18 – 170	na[c]	na[c]
Dibenzo(a,h)anthracene	436/828	53	1.6 J	1,500	60	1.0 – 2,000	na[c]	na[c]
Dibenzofuran	248/827	30	1.0 J	4,200	50	1.7 – 2,000	na[c]	na[c]
Fluoranthene	797/828	96	18	24,000	900	19 – 340	na[c]	na[c]
Fluorene	382/828	46	1.4 J	6,800	80	1.8 – 2,000	na[c]	na[c]
Indeno(1,2,3-cd)pyrene	726/823	88	6.5	4,300	200	6.4 – 1,600	na[c]	na[c]
Naphthalene	152/818	19	3.0 J	5,300	nc	1.0 – 2,000	na[c]	na[c]
Phenanthrene	759/828	92	7.1	28,000	400	18 – 200	na[c]	na[c]
Pyrene	788/828	95	19	16,000	700	18 – 170	na[c]	na[c]
Total HPAH[d]	803/828	97	20	85,000	4,000	nc	na[c]	na[c]
Total LPAH[e]	763/828	92	9.1	44,000	700	nc	na[c]	na[c]
Total PAH	805/828	97	20	128,000	4,000	nc	nv	nv
Phthalates[f]								
BBP	425/822	52	2.0	7,100	80	1.8 – 2,000	na[c]	na[c]
Dimethyl phthalate	156/822	19	2.0 J	200	nc	1.8 – 2,000	na[c]	na[c]
Chlorobenzenes								
1,2,4-Trichlorobenzene	5/816	1	1.6 J	72 J	nc	0.40 – 2,000	na[c]	na[c]
1,2-Dichlorobenzene	17/816	2	1.3 J	520 J	nc	0.40 – 2,000	na[c]	na[c]
1,4-Dichlorobenzene	39/816	5	1.7 J	1,600 J	nc	0.20 – 2,000	na[c]	na[c]
Hexachlorobenzene	46/819	6	0.4 J	95 J	nc	0.11 – 2,000	na[c]	na[c]
Other SVOCs								
2,4-Dimethylphenol	5/813	1	6.1	290 J	nc	6.0 – 2,000	29	29
4-Methylphenol	82/831	10	4.8 J	4,600 J	nc	8.6 – 2,000	670	670
Benzoic acid	70/822	9	54 J	4,500	nc	13 – 3,000	650	650
Benzyl alcohol	15/812	2	8.2 J	670	nc	9.2 – 690	57	73
n-Nitrosodiphenylamine	23/818	3	6.5	230	nc	1.8 – 2,000	na[c]	na[c]
Pentachlorophenol	12/785	2	14 J	410	nc	7.6 – 4,900	360	690
Phenol	257/831	31	10 J	2,800	90	7.3 – 790	420	1,200

Note: An SVOC was included in this table if it exceeded the SQS or if it was identified as a COC in the HHRA (Appendix B) based on a sediment exposure pathway.

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results. Mean concentrations were not calculated if the number of samples was ≤ 20 or the detection frequency was ≤ 25% because it is uncertain if these means would represent LDW-wide conditions.

[b] RLs are based only on non-detect samples.

[c] SMS criteria are in mg/kg OC for these chemicals; OC-normalized concentrations and criteria for these chemicals are presented in Table 4-77.

[d] Total HPAHs were calculated as the sum of benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, total benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, and pyrene.

[e] Total LPAHs were calculated as the sum of acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene.

[f] BEHP data are summarized in Table 4-56 in Section 4.2.6.1.

BBP – butyl benzyl phthalate
CSL – cleanup screening level
dw – dry weight
na – not applicable
nc – not calculated
nd – not detected
nv – no value
J – estimated concentration

HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
OC – organic carbon
PAH – polycyclic aromatic hydrocarbon
RL – reporting limit
SVOC – semivolatile organic compound
SMS – Washington State Sediment Management Standards
SQS – sediment quality standards

Table 4-77. Summary of OC-normalized surface sediment data for selected SVOCs with OC-normalized SMS criteria

	Detection Frequency		Concentration (mg/kg OC)[a]				SMS Criteria (mg/kg OC)	
Chemical	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	Range of RLs[c]	SQS	CSL
PAHs								
2-Methylnaphthalene	123/753	16	0.090	160	nc	0.059 – 94	38	64
Acenaphthene	283/760	37	0.064 J	260	3	0.059 – 94	16	57
Anthracene	548/760	72	0.10	380	7	0.65 – 94	220	1,200
Benzo(a)anthracene	712/760	94	0.44	440	20	0.29 – 34	110	270
Benzo(a)pyrene	710/756	94	0.25	420	20	0.29 – 68	99	210
Benzo(g,h,i)perylene	648/755	86	0.22	180	8	0.65 – 94	31	78
Total benzofluoranthenes	715/754	95	0.49 J	890	40	nc	230	450
Chrysene	731/760	96	0.86	410	20	0.67 – 34	110	460
Dibenzo(a,h)anthracene	413/760	54	0.080	71	3	0.034 – 170	12	33
Dibenzofuran	232/759	31	0.10 J	220	3	0.059 – 94	15	58
Fluoranthene	750/760	99	0.92	1,300	50	0.67 – 68	160	1,200
Fluorene	357/760	47	0.090	290	4.0	0.059 – 94	23	79
Indeno(1,2,3-cd)pyrene	689/755	91	0.23	200	9	0.29 – 59	34	88
Naphthalene	135/753	18	0.13 J	260	2.3	0.059 – 94	99	170
Phenanthrene	719/760	95	0.36	1,500	20	0.67 – 34	100	480
Pyrene	743/760	98	0.87	840	40	0.67 – 34	1,000	1,400
Total HPAH[d]	754/760	99	1.6 J	4,500	200	nc	960	5,300
Total LPAH[e]	721/760	95	0.46	2,300	30	nc	370	780
Phthalates[f]								
BBP	406/757	54	0.071	530 J	4	0.060 – 94	4.9	64

CHEMICAL	DETECTION FREQUENCY		CONCENTRATION (mg/kg OC)[a]				SMS CRITERIA (mg/kg OC)	
	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	CALCULATED MEAN[B]	RANGE OF RLs[C]	SQS	CSL
Dimethyl phthalate	149/757	20	0.14 J	15	nc	0.059 – 94	53	53
Chlorobenzenes								
1,2,4-Trichlorobenzene	4/751	1	0.057 J	0.61 J	nc	0.020 – 94	0.81	1.8
1,2-Dichlorobenzene	15/751	2	0.068 J	2.9 J	nc	0.020 – 94	2.3	2.3
1,4-Dichlorobenzene	34/751	5	0.097 J	65	nc	0.023 – 94	3.1	9
Hexachlorobenzene	45/753	6	0.02 J	3.8	nc	0.0095 – 94	0.38	2.3
Other SVOCs								
n-Nitrosodiphenylamine	21/753	3	0.23	7.9	nc	0.059 – 94	11	11

[a] Summary statistics for OC-normalized concentrations were calculated using only samples with TOC contents ≥ 0.5% and ≤ 4.0%. At very low or very high TOC contents, OC normalization is not appropriate for comparison to SMS (Michelsen and Bragdon-Cook 1993).

[b] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results. Mean concentrations were not calculated if the number of samples was ≤ 20 or the detection frequency was ≤ 25% because it is uncertain if these means would represent LDW-wide conditions.

[c] RLs are based only on non-detect samples.

[d] Total HPAHs were calculated as the sum of benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, total benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, and pyrene.

[e] Total LPAHs were calculated as the sum of acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene.

[f] BEHP data are summarized in Table 4-58 in Section 4.2.6.1.

BBP – butyl benzyl phthalate
CSL – cleanup screening level
J – estimated concentration
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
nc – not calculated
OC – organic carbon
PAH – polycyclic aromatic hydrocarbon
RL – reporting limit
SVOC – semivolatile organic compound
SMS – Washington State Sediment Management Standards
SQS – sediment quality standards

At least one individual PAH listed in Table 4-76 was detected in samples from 97% of the locations. The individual PAHs with the highest mean concentrations were fluoranthene (900 µg/kg dw), pyrene (700 µg/kg dw), chrysene (500 µg/kg dw), and phenanthrene (400 µg/kg dw). Concentrations of total high-molecular-weight polycyclic aromatic hydrocarbons (HPAHs) were generally higher than those of total LPAHs (means of 4,000 and 700 µg/kg, respectively).

The maximum total HPAH concentration (85,000 µg/kg dw) was detected at a location near RM 3.6 in the T-117 EAA (T117-SE-37-G). Other areas with high total HPAH concentrations were: 1) west of the navigation channel near RM 0.6 (51,000 µg/kg dw at DR044), 2) at the RM 3.8 EAA on the eastern shoreline (51,000 µg/kg dw at R23), 3) near RM 2.7 on the western shoreline (48,000 µg/kg dw at LDW-SS95), and 4) near RM 1.0 on the eastern shoreline (42,000 µg/kg dw at LDW-SS35). HPAH concentrations exceeded the SQS but not the CSL in 24 samples (2.9%) and exceeded the CSL in 4 samples (0.48%) (Table 4-16). Approximately half of the exceedances were located within EAAs (Map 4-61).

The maximum total LPAH concentration (44,000 µg/kg dw) was detected at a location near RM 2.7 on the western shoreline (LDW-SS95). Other areas with high total LPAH concentrations were: 1) in the T-117 EAA (43,000 µg/kg dw at T117-SE-37-G), 2) near RM 1.0 on the eastern shoreline (34,000 µg/kg dw at LDW-SS35), and 3) in the Boeing Plant 2/Jorgensen Forge EAA at RM 3.6 (10,900 µg/kg dw at SD-307-S). Several other areas also had total HPAH and LPAH concentrations greater than the numerical 95th percentile, as shown on Maps 4-61 and 4-62. Total LPAH concentrations exceeded the SQS but not the CSL in three samples and exceeded the CSL in three samples, both inside and outside of EAAs (Table 4-16; Map 4-62).

Of the two phthalate compounds presented in Table 4-76, BBP was detected more frequently than dimethyl phthalate (in 52 and 19% of samples, respectively). Most of locations with high BBP concentrations in surface sediment were located in the vicinity of the Duwamish/Diagonal EAA (Map 4-63), as were the highest dimethyl phthalate concentrations.

BBP concentrations exceeded the SQS but not the CSL in 71 samples (8.6%) and exceeded the CSL in 8 samples (0.97%) (Table 4-16). Most of the BBP SQS and CSL exceedances were located within EAAs, although six SQS exceedances were located south of the RM 3.8 EAA, and several other isolated SQS exceedances were located at various areas in the LDW. Approximately half of the exceedances were located between RM 0.4 and RM 0.6 in and around the Duwamish/Diagonal EAA (Map 4-63).

Each of the four chlorobenzenes listed in Table 4-76 was detected in 6% or fewer of the surface sediment samples. Samples with detected concentrations of 1,2-dichlorobenzene, 1,4-dichlorobenzene, and 1,2,4-trichlorobenzene were collected primarily between RM 0.4 and RM 0.6 on the east side of the LDW in the vicinity of the Duwamish/Diagonal EAA. The highest hexachlorobenzene concentrations (95 and 63 µg/kg dw) were detected near RM 1.8 west of the navigation channel (LDW-SS68) and between RM 3.0 and RM 3.1 on the western shoreline (B7a), respectively. All other detected hexachlorobenzene concentrations were less than 30 µg/kg dw.

Among the seven chemicals in the “other SVOC” category in Table 4-76, phenol was detected most frequently (in 31% of the samples). The highest concentrations of phenol were primarily located between RM 3.5 and RM 3.7 in the Boeing Plant 2/Jorgensen Forge EAA, near RM 3.6 in the T117 EAA, and in Slip 6 (Map 4-64). Phenol concentrations exceeded the SQS but not the CSL in 17 samples (2.0%) and exceeded the CSL in 6 samples (0.72%) (Table 4-16). Three of the phenol CSL exceedances were located in the T-117 EAA, and two were in Slip 6. Most of the remaining phenol SQS and CSL exceedances were located within the Boeing Plant 2/Jorgensen Forge EAA, although there were also several other isolated SQS exceedances at various locations in the LDW (Map 4-64).

The highest concentrations of benzoic acid were located primarily between RM 4.0 and RM 4.2 and in Slip 6 (Map 4-65). Five of the eight benzoic acid CSL exceedances were located in this area (Map 4-65).

4.2.9.2 Subsurface sediment

Detection frequencies and ranges of concentrations of SVOCs in all subsurface sediment samples collected from any interval are presented in Table 4-78. The most frequently detected SVOCs in subsurface sediment were PAH compounds; individual PAHs were detected in 15 to 88% of all samples. Other SVOCs presented in Table 4-78 that were most frequently detected were BBP and benzoic acid in 52 and 40% of all samples, respectively.

Table 4-78. Summary of subsurface sediment data from any interval for selected SVOCs

CHEMICAL	DETECTION FREQUENCY[a] RATIO	%	CONCENTRATION (µg/kg dw)[b] MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLs[c]
PAHs					
2-Methylnaphthalene	43/281	15	2.9 J	4,500	1.6 – 400
Acenaphthene	94/304	31	1.3 J	4,600	4.30 – 400
Anthracene	200/304	66	4.3 J	1,900	4.30 – 400
Benzo(a)anthracene	243/304	80	12 J	4,500	4.30 – 180
Benzo(a)pyrene	237/304	78	11 J	5,300	4.30 – 400
Benzo(g,h,i)perylene	207/298	69	11 J	1,000	4.30 – 400
Total benzofluoranthenes	248/304	82	21 J	10,200	nc
Chrysene	247/304	81	12 J	7,200	4.30 – 180
Dibenzo(a,h)anthracene	119/304	39	3.2 J	660 J	4.30 – 910
Dibenzofuran	58/304	19	2.9 J	1,700	1.7 – 400
Fluoranthene	262/304	86	12 J	13,000	4.30 – 340
Fluorene	104/304	34	4.9 J	4,300	2.3 – 400
Indeno(1,2,3-cd)pyrene	212/298	71	10 J	1,500	4.30 – 400
Naphthalene	70/293	24	1.9 J	3,400	1.7 – 400
Phenanthrene	244/304	80	10 J	13,000	4.30 – 400
Pyrene	266/304	88	9.9 J	10,000	4.30 – 170
Total HPAH[d]	268/304	88	9.9 J	47,000	nc
Total LPAH[e]	244/304	80	12.00	27,000 J	nc
Total PAH	269/304	88	9.9 J	57,000 J	nc
Phthalates[f]					
BBP	147/283	52	5.6 J	610	5.8 – 400
Dimethyl phthalate	20/283	7	4.2 J	8,800	2.4 – 400

	Detection Frequency[a]		Concentration (µg/kg dw)[b]		
Chemical	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[c]
Chlorobenzenes					
1,2,4-Trichlorobenzene	27/293	9	3.6 J	110 J	0.87 – 400
1,2-Dichlorobenzene	31/292	11	0.98 J	160	0.78 – 400
1,4-Dichlorobenzene	56/292	19	2.1 J	750 J	0.86 – 400
Hexachlorobenzene	3/280	1	4.6 J	10	0.78 – 400
Other SVOCs					
2,4-Dimethylphenol	21/281	7	3.7 J	46	5.8 – 790
4-Methylphenol	15/304	5	8.6 J	110 J	3.8 – 400
Benzoic acid	122/304	40	35 J	3,000 J	58 – 4,000
Benzyl alcohol	32/281	11	7.7 J	210	4.9 – 2,000
n-Nitrosodiphenylamine	1/281	< 1	33	33	2.9 – 7,300
Pentachlorophenol	54/304	18	16 J	930 J	12 – 3,100
Phenol	57/304	19	12 J	3,100	4.4 – 790

Note: Data included in the subsurface dataset are described in Section 4.1.2.2; subsurface sediment samples collected from areas that were subsequently dredged are not summarized in this table because they do not represent existing conditions. An SVOC was included in this table if it exceeded the SQS in at least one surface sediment sample or if it was identified as a COC in the HHRA (Appendix B) based on a sediment exposure pathway.

[a] Subsurface sediment samples collected from areas that were subsequently dredged are not included in this table

.[b] Mean concentrations were not calculated because sampling was generally focused in contaminated areas, resulting in a non-random sampling pattern, which is not likely representative of the entire LDW.

[c] RLs are based only on non-detect samples.

[d] Total HPAHs were calculated as the sum of benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, total benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, and pyrene.

[e] Total LPAHs were calculated as the sum of acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene.

[f] BEHP data are summarized in Table 4-61 in Section 4.2.6.2.

BBP – butyl benzyl phthalate
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
J – estimated concentration
nc – not calculated
PAH – polycyclic aromatic hydrocarbon
RL – reporting limit
SVOC – semivolatile organic compound
U – not detected at RL shown

Detection frequencies and ranges of concentrations of SVOCs in subsurface sediment collected from specific 1- and 2-ft intervals[98] within the top 10 ft of sediment are presented in Table 4-79. Over 90% of the samples represented in these tables were collected from the top 4 ft of sediment. Relatively few samples were analyzed from

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[98] Subsurface sediment cores were sampled at various depth intervals during different sampling events. To calculate summary statistics, data were categorized according to specified depth intervals, as described for total PCBs in Section 4.2.2.2.

depths greater than 4 ft below mudline; most of these samples were collected and analyzed as part of the RI at locations with SMS exceedances of SVOCs in upper intervals.

Table 4-79. Summary of subsurface sediment data for selected SVOCs in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment

Chemical	Sampling Interval (ft)[a]	Detection Frequency		Concentration (µg/kg dw)[b]		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[c]
PAHs						
Total HPAH[d] – 1-ft intervals	0 to 1	60/64	94	122 J	34,700	nc
	1 to 2	38/45	84	13 J	40,000	nc
	2 to 3	7/10	70	115 J	4,400	nc
	3 to 4	4/9	44	21 J	323 J	nc
	4 to 5	1/1	100	580 J	580 J	nc
Total HPAH[d] – 2-ft intervals	0 to 2	72/74	97	80	27,900	nc
	2 to 4	62/73	85	9.9 J	47,000	nc
	4 to 6	10/10	100	140	22,000	nc
	6 to 8	5/5	100	60 J	38,000 J	nc
	8 to 10	5/5	100	58.0	2,040	nc
Total LPAH[e] – 1-ft intervals	0 to 1	54/64	84	12 J	2,100 J	nc
	1 to 2	34/45	76	24 J	7,500	nc
	2 to 3	4/10	40	20	2,630 J	nc
	3 to 4	2/9	22	43 J	220 J	nc
	4 to 5	1/1	100	47 J	47 J	nc
Total LPAH[e] – 2-ft intervals	0 to 2	72/74	97	15 J	3,800 J	nc
	2 to 4	56/73	77	20	27,000 J	nc
	4 to 6	9/10	90	141 J	3,900	nc
	6 to 8	4/5	80	770 J	9,800	nc
	8 to 10	5/5	100	12.00	275 J	nc
Total PAH – 1-ft intervals	0 to 1	61/64	95	24	36,200	nc
	1 to 2	38/45	84	13 J	42,600 J	nc
	2 to 3	7/10	70	135 J	5,000 J	nc
	3 to 4	4/9	44	21 J	366 J	nc
	4 to 5	1/1	100	630 J	630 J	nc
Total PAH – 2-ft intervals	0 to 2	73/74	99	24	29,500 J	nc
	2 to 4	62/73	85	9.9 J	57,000 J	nc
	4 to 6	10/10	100	281 J	23,000 J	nc
	6 to 8	5/5	100	60 J	46,000 J	nc
	8 to 10	5/5	100	70.0	2,210	nc
Phthalates[f]						
BBP– 1-ft intervals	0 to 1	33/41	80	5.9	610	5.9 – 180
	1 to 2	22/45	49	14	400	5.8 – 36

Table 4-79, cont. **Summary of subsurface sediment data for selected SVOCs in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment**

Chemical	Sampling Interval (ft)[a]	Detection Frequency		Concentration (µg/kg dw)[b]		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[c]
	2 to 3	3/10	30	6.6	10	5.9 – 400
	3 to 4	2/9	22	6.4	7.7	5.8 – 20
	4 to 5	1/1	100	18	18	na
BBP – 2-ft intervals	0 to 2	51/74	69	5.9	610	5.8 – 110
	2 to 4	34/73	47	5.8 J	180	5.8 – 42
	4 to 6	5/10	50	12 J	48	6.5 – 36
	6 to 8	4/5	80	15 J	35 J	6.5
	8 to 10	0/5	0	nd	nd	6.1 – 20.0
Dimethyl phthalate – 1-ft intervals	0 to 1	3/41	7	13 J	1,700	13 – 180
	1 to 2	1/45	2	16 J	16 J	12 – 130
	2 to 3	0/10	0	nd	nd	14 – 400
	3 to 4	0/9	0	nd	nd	14 – 20
	4 to 5	0/1	0	nd	nd	6.6
Dimethyl phthalate – 2-ft intervals	0 to 2	5/74	7	9.9 J	84	12 – 110
	2 to 4	4/73	5	30	8,800	14 – 140
	4 to 6	2/10	20	20	210	6.5 – 44
	6 to 8	3/5	60	16	69 J	6.5
	8 to 10	0/5	0	nd	nd	6.6 – 62
Chlorobenzenes						
1,2,4-Tri-chlorobenzene – 1-ft intervals	0 to 1	5/42	12	3.6 J	18 J	5.8 – 180
	1 to 2	6/46	13	4.1 J	17 J	5.8 – 20.0
	2 to 3	0/11	0	nd	nd	5.9 – 400
	3 to 4	0/10	0	nd	nd	5.8 – 20
	4 to 5	0/2	0	nd	nd	6.6 – 8.3
	5 to 6	0/1	0	nd	nd	8.3
	6 to 7	0/1	0	nd	nd	8.2
	7 to 8	0/1	0	nd	nd	9.5
	8 to 9	0/1	0	nd	nd	9.5
	9 to 10	0/1	0	nd	nd	8.4
1,2,4-Tri-chlorobenzene – 2-ft intervals	0 to 2	11/76	14	3.9 J	13 J	5.8 – 110
	2 to 4	4/74	5	4.1 J	110 J	5.8 – 25.0
	4 to 6	2/11	18	11	18	6.5 – 20.0
	6 to 8	3/6	50	9.8	14	6.5 – 8.2
	8 to 10	0/5	0	nd	nd	5.40 – 6.6

Table 4-79, cont. **Summary of subsurface sediment data for selected SVOCs in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment**

CHEMICAL	SAMPLING INTERVAL (ft)[a]	DETECTION FREQUENCY		CONCENTRATION (µg/kg dw)[b]		
		RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLS[c]
1,2-Dichlorobenzene – 1-ft intervals	0 to 1	4/42	10	1.7 J	17	2.0 – 180
	1 to 2	4/46	9	2.9 J	9.6	0.78 – 20.0
	2 to 3	0/11	0	nd	nd	0.86 – 400
	3 to 4	0/10	0	nd	nd	0.86 – 6.0
	4 to 5	1/2	50	10	10	1.7
	5 to 6	0/1	0	nd	nd	1.7
	6 to 7	0/1	0	nd	nd	1.6
	7 to 8	0/1	0	nd	nd	1.9
	8 to 9	0/1	0	nd	nd	1.9
	9 to 10	0/1	0	nd	nd	1.7
1,2-Dichlorobenzene – 2-ft intervals	0 to 2	8/76	11	1.7 J	20	1.7 – 110
	2 to 4	8/74	11	3.6 J	150	0.86 – 42
	4 to 6	2/11	18	10	12	1.7 – 20.0
	6 to 8	3/6	50	12	160	1.6 – 6.6
	8 to 10	0/5	0	nd	nd	1.10 – 6.6
1,4-Dichlorobenzene – 1-ft intervals	0 to 1	15/42	36	3.0 J	750 J	2.0 – 180
	1 to 2	9/46	20	3.5 J	17 J	1.7 – 20
	2 to 3	2/11	18	2.1 J	3.0 J	1.8 – 400
	3 to 4	0/10	0	nd	nd	0.86 – 6.0
	4 to 5	1/2	50	7.9	7.9	1.7
	5 to 6	0/1	0	nd	nd	1.7
	6 to 7	0/1	0	nd	nd	1.6
	7 to 8	0/1	0	nd	nd	1.9
	8 to 9	0/1	0	nd	nd	1.9
	9 to 10	0/1	0	nd	nd	1.7
1,4-Dichlorobenzene – 2-ft intervals	0 to 2	16/76	21	3.0 J	380 J	1.7 – 110
	2 to 4	13/74	18	2.1 J	38	1.6 – 42
	4 to 6	7/11	64	3.9 J	31	1.7 – 20.0
	6 to 8	4/6	67	4.0 J	24	1.6 – 6.5
	8 to 10	0/5	0	nd	nd	1.10 – 6.6
Hexachlorobenzene – 1-ft intervals	0 to 1	1/41	2	5.9	5.9	0.83 – 180
	1 to 2	1/45	2	10	10	0.78 – 20.0
	2 to 3	0/10	0	nd	nd	0.86 – 400
	3 to 4	0/9	0	nd	nd	0.86 – 6.0
	4 to 5	0/1	0	nd	nd	6.6

Table 4-79, cont. Summary of subsurface sediment data for selected SVOCs in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment

Chemical	Sampling Interval (ft)[a]	Detection Frequency		Concentration (µg/kg dw)[b]		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[c]
Hexachlorobenzene – 2-ft intervals	0 to 2	2/74	3	5.9	10	0.78 – 110
	2 to 4	0/73	0	nd	nd	0.86 – 42
	4 to 6	0/10	0	nd	nd	6.5 – 20.0
	6 to 8	1/5	20	4.6 J	4.6 J	6.5 – 6.6
	8 to 10	0/5	0	nd	nd	6.1 – 20.0
Other SVOCs						
2,4-Dimethylphenol – 1-ft intervals	0 to 1	2/41	5	25 J	27 J	5.8 – 350
	1 to 2	3/45	7	9.5 J	14 J	5.8 – 30
	2 to 3	0/10	0	nd	nd	5.9 – 790
	3 to 4	0/9	0	nd	nd	5.8 – 34
	4 to 5	0/1	0	nd	nd	6.6
2,4-Dimethylphenol – 2-ft intervals	0 to 2	5/74	7	9.5 J	27 J	5.8 – 330
	2 to 4	5/73	7	6.3 J	46	5.8 – 42
	4 to 6	3/10	30	6.5 J	9.2	6.5 – 20.0
	6 to 8	4/5	80	8.5 J	24 J	6.5
	8 to 10	3/5	60	3.7 J	10	6.6 – 20.0
4-Methylphenol – 1-ft intervals	0 to 1	2/64	3	17 J	42 J	20 – 310
	1 to 2	1/45	2	24	24	19 – 130
	2 to 3	0/10	0	nd	nd	20 – 400
	3 to 4	0/9	0	nd	nd	19 – 34
	4 to 5	0/1	0	nd	nd	66
4-Methylphenol – 2-ft intervals	0 to 2	3/74	4	13 J	42 J	20 – 110
	2 to 4	3/73	4	23	110 J	19 – 140
	4 to 6	1/10	10	42 J	42 J	20.0 – 66
	6 to 8	2/5	40	37 J	48 J	65 – 66
	8 to 10	0/5	0	nd	nd	20.0 – 66
Benzoic acid – 1-ft intervals	0 to 1	40/64	62	52 J	2,000 J	59 – 2,700
	1 to 2	23/45	51	48 J	450 J	59 – 200
	2 to 3	5/10	50	66	110 J	100 – 4,000
	3 to 4	6/9	67	35 J	130	58 – 140
	4 to 5	0/1	0	nd	nd	590
Benzoic acid – 2-ft intervals	0 to 2	41/74	55	50 J	490	58 – 1,100
	2 to 4	32/73	44	54 J	3,000 J	58 – 350
	4 to 6	0/10	0	nd	nd	200 – 590
	6 to 8	1/5	20	320 J	320 J	590
	8 to 10	0/5	0	nd	nd	200 – 620

Table 4-79, cont. Summary of subsurface sediment data for selected SVOCs in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment

Chemical	Sampling Interval (ft)[a]	Detection Frequency		Concentration (µg/kg dw)[b]		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[c]
Benzyl alcohol – 1-ft intervals	0 to 1	10/41	24	18 J	200	29 – 890
	1 to 2	7/45	16	19 J	210	19 – 99
	2 to 3	0/10	0	nd	nd	29 – 2,000
	3 to 4	0/9	0	nd	nd	29 – 34
	4 to 5	0/1	0	nd	nd	33
Benzyl alcohol – 2-ft intervals	0 to 2	11/74	15	18 J	200	19.0 – 550
	2 to 4	7/73	10	20 J	34 J	20.0 – 210
	4 to 6	1/10	10	52	52	20.0 – 33
	6 to 8	0/5	0	nd	nd	33
	8 to 10	0/5	0	nd	nd	20.0 – 38
n-Nitroso-diphenylamine – 1-ft intervals	0 to 1	1/41	2	33	33	8.2 – 320
	1 to 2	0/45	0	nd	nd	5.8 – 620
	2 to 3	0/10	0	nd	nd	5.9 – 400
	3 to 4	0/9	0	nd	nd	5.8 – 34
	4 to 5	0/1	0	nd	nd	32
n-Nitroso-diphenylamine – 2-ft intervals	0 to 2	1/74	1	33	33	6 – 870
	2 to 4	0/73	0	nd	nd	5.8 – 7,300
	4 to 6	0/10	0	nd	nd	20.0 – 510
	6 to 8	0/5	0	nd	nd	6.5 – 2,600
	8 to 10	0/5	0	nd	nd	20.0 – 65
Pentachlorophenol – 1-ft intervals	0 to 1	15/64	23	16 J	930 J	29 – 3,100
	1 to 2	8/45	18	17 J	120 J	29 – 100
	2 to 3	1/10	10	26 J	26 J	29 – 2,000
	3 to 4	0/9	0	nd	nd	29 – 34
	4 to 5	0/1	0	nd	nd	33
Pentachlorophenol – 2-ft intervals	0 to 2	17/74	23	17 J	730	29 – 550
	2 to 4	10/73	14	18 J	190	29 – 180
	4 to 6	9/10	90	36	63	100
	6 to 8	4/5	80	45	800	33
	8 to 10	1/5	20	19 J	19 J	30 – 98.00
Phenol – 1-ft intervals	0 to 1	17/64	27	14 J	3,100	20 – 350
	1 to 2	9/45	20	15 J	150	19 – 130
	2 to 3	1/10	10	13 J	13 J	20 – 790
	3 to 4	2/9	22	13 J	13 J	19 – 140
	4 to 5	0/1	0	nd	nd	66

Table 4-79, cont. Summary of subsurface sediment data for selected SVOCs in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment

CHEMICAL	SAMPLING INTERVAL (ft)[a]	DETECTION FREQUENCY		CONCENTRATION (µg/kg dw)[b]		
		RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLS[c]
Phenol – 2-ft intervals	0 to 2	20/74	27	14 J	180	19 – 220
	2 to 4	17/73	23	13 J	110	19 – 140
	4 to 6	0/10	0	nd	nd	20.0 – 73
	6 to 8	0/5	0	nd	nd	65 – 66
	8 to 10	0/5	0	nd	nd	20.0 – 66

Note: Data included in the subsurface dataset are described in Section 4.1.2.2; subsurface sediment samples collected from areas that were subsequently dredged are not summarized in this table because they do not represent existing conditions.An SVOC was included in this table if it exceeded the SQS or if it was identified as a COC in the HHRA (Appendix B) based on a sediment exposure pathway.

[a] For the calculation of summary statistics in this table, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to the 1- or 2-ft sampling category that best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were averaged to obtain a calculated concentration for the corresponding 2-ft interval. Some of the subsurface data were collected from intervals other than 1 or 2 ft; therefore, the detection frequency ratio for "any interval" in Table 4-78 may reflect data for intervals not included in these categories.

[b] Mean concentrations were not calculated because sampling was generally focused in contaminated areas, resulting in a non-random sampling pattern, which is not likely representative of the entire LDW.

[c] RLs are based only on non-detect samples.

[d] Total HPAHs were calculated as the sum of benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, total benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, and pyrene.

[e] Total LPAHs were calculated as the sum of acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene.

[f] BEHP data are summarized in Table 4-61 in Section 4.2.6.2.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
J – estimated concentration
na – not applicable
nc – not calculated
nd – not detected
PAH – polycyclic aromatic hydrocarbon
RL – reporting limit
SVOC – semivolatile organic compound

PAH data in 1- and 2-ft subsurface intervals for the top 6 ft of sediment are shown on Maps 4-66a through 4-66c (total HPAHs) and Maps 4-67a through 4-67c (total LPAHs). The maximum total HPAH concentration (64,000 µg/kg dw) was detected at a location near RM 1.2 on the eastern shoreline (LDW-SC23) at an interval of 3 to 3.5 ft. Other areas with high total HPAH concentrations include:

- Slip 3 (LDW-SC37) (47,000 µg/kg dw in the 2-to-4-ft interval)
- Near RM 2.7 on the western shoreline (C3; within an area that was subsequently dredged) (41,000 µg/kg dw in the 0-to-3.3-ft interval)
- Near RM 1.4 on the west side of the navigation channel (LDW-SC26) (38,000 µg/kg dw in the 6-to-8-ft interval)
- Near RM 0.5 on the western shoreline (LDW-SC11) (34,700 µg/kg dw in the 0-to-1-ft interval)

The maximum total LPAH concentration (27,000 µg/kg dw) was detected in Slip 1 (LDW-SC17) at an interval of 2 to 4 ft. High total HPAH concentrations were also reported in the following areas:

- Near RM 1.2 on the east side of the navigation channel (LDW-SC23) (25,000 µg/kg dw in the 3-to-3.5-ft interval)
- Slip 3 (LDW-SC37) (10,500 µg/kg dw in the 2-to-4-ft interval)
- Near RM 2.7 on the western shoreline (C3; within an area that was subsequently dredged) (10,000 µg/kg dw in the 0-to-3.3-ft interval)
- Near RM 1.4 on the west side of the navigation channel (LDW-SC26) (8,500 µg/kg dw in the 6-to-8-ft interval)

Of the SVOCs described in this section, BBP and benzoic acid were the most frequently detected in subsurface sediment samples after PAHs (Table 4-78). Most of the locations with the highest BBP concentrations in any subsurface interval were located between RM 0.4 and RM 0.6, in the vicinity of the Duwamish/Diagonal EAA (similar to BEHP, as described in Section 4.2.6.2). The highest BBP concentration in this area was 1,300 µg/kg dw in the 2-to-2.5-ft interval at DUD006. The highest benzoic acid concentration in subsurface sediment was detected in a sample from Slip 1 (3,000 µg/kg in the 2-to-4-ft interval at LDW-SS17). Benzoic acid concentrations ranging from 1,200 to 2,000 µg/kg were in detected in 11 samples collected in the vicinity of RM 4.2.

Chlorobenzenes were detected in 1 to 19% of all subsurface samples in which they were analyzed from any interval (Table 4-78). Of the other SVOCs listed in Table 4-78, benzoic acid was the most frequently detected (in 40% of all subsurface samples) and had the highest mean concentration (200 µg/kg dw). Other SVOCs were detected in 17% or less of all subsurface samples in which they were analyzed from any interval.

4.2.9.3 Tissue

Table 4-80 presents the detection frequency and range of SVOC concentrations in tissue.[99] Of the SVOCs listed in Table 4-80, PAHs, benzoic acid, and phenol were the only compounds detected in 17% or more of the composite tissue samples.

Table 4-80. Summary of data for selected SVOCs in tissue samples

Chemical	Detection Frequency		Concentration (µg/kg ww)		
	Ratio	%	Minimum Detect	Maximum Detect	Range of RLs[a]
PAHs					
Total HPAH[b]	180/235	77	0.21 J	3,200 J	nc
Total LPAH[c]	150/235	64	0.094 J	450 J	nc
Total PAH	187/235	80	0.43 J	3,630 J	nc
Phthalates[d]					
BBP	25/230	11	190 J	2,800	9 – 4,000
Dimethyl phthalate	6/235	3	7.6 J	420 JM	3.6 – 2,900
Chlorobenzenes					
Hexachlorobenzene	23/235	10	0.38 JN	6.6 JN	0.8 – 26
Other SVOCs					
2,4-Dimethylphenol	2/235	1	71 J	210 J	3.6 – 1,200
4-Methylphenol	20/235	9	15 J	44,000	3.6 – 1,200
Benzoic acid	154/235	66	340 J	54,000	36 – 12,000
Benzyl alcohol	44/235	19	12 J	3,450	3.6 – 1,200
Carbazole	2/235	1	6,000	14,000	3.6 – 2,900
n-Nitrosodiphenylamine	1/235	<1	170 J	170 J	3.6 – 2,900
Pentachlorophenol	16/235	7	1.1 J	4,700	1.7 – 5,800
Phenol	40/235	17	17 J	2,200	3.6 – 1,500

Note: An SVOC is included in this table if it was detected in any tissue sample and was identified as a chemical of potential concern in either the HHRA (Appendix B) based on seafood consumption or in the ERA (Appendix A) based on a tissue exposure pathway.

[a] RLs are based only on non-detect samples.

[b] Total HPAHs were calculated as the sum of benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, total benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, and pyrene.

[c] Total LPAHs were calculated as the sum of acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene.

[d] BEHP data are summarized in Table 4-63 in in Section 4.2.6.4.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
J – estimated concentration
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
M – calculated whole-body concentration
N – tentative identification
nc – not calculated
SVOC – semivolatile organic compound
ww – wet weight

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[99] An SVOC is discussed in this section if a LOAEL-based HQ was greater than or equal to 1.0 for any receptor in the ERA, if it was identified as a COC in the HHRA based on seafood consumption, or if it was detected in any tissue sample.

In general, individual PAHs were detected frequently in all tissue types except Pacific staghorn sculpin (in only 3 of the 24 samples), English sole fillets without skin (in none of the 6 samples), and amphipods (in only 1 of the 4 samples) (Table 4-81). Although the same analytical method was used to analyze the Pacific staghorn sculpin samples as was used for the other tissue samples collected in 2004, the RLs for PAHs and phthalates were high for Pacific staghorn sculpin samples because analytical dilutions were used to minimize matrix interferences at the discretion of the laboratory to reduce the potential impact to their analytical equipment. The dilutions resulted in reduced sensitivity of the analysis (i.e., higher RLs) for this subset of samples. cPAH concentrations in tissue are discussed in Section 4.2.5.4. Benzoic acid, which, after PAHs, was the next most frequently detected SVOC in tissue, was detected frequently in fish, benthic invertebrates, mussels, and clams but was not detected in any crab samples (Table 4-81).

RLs for pentachlorophenol (PCP) were elevated (up to 5,800 µg/kg ww) for some of the English sole, Pacific staghorn sculpin, shiner surfperch, and benthic invertebrate samples because of analytical dilutions. Forty-nine samples of various tissue types with RLs of 5,700 to 5,800 µg/kg ww were re-analyzed using a different analytical method to achieve lower RLs. PCP was detected in six of the re-analyzed tissue samples at low concentrations ranging from 1.1 to 2.8 µg/kg ww; RLs for the re-analyses ranged from 3.3 to 11 µg/kg ww. In the original analyses, PCP was detected in two shiner surfperch whole-body samples at 2,200 µg/kg ww. Neither of these detected results were confirmed by the re-analysis; both samples had non-detected concentrations with RLs of 4.5 and 4.6 µg/kg ww. The results from the re-analysis were selected as the final result because of the enhanced sensitivity of the analytical method.

Table 4-81. Summary of detection frequencies for SVOCs in tissue samples

Chemical	Detection Frequency (%)																	
	Fish – Whole Body				Fish – Fillet					Crab – EM		Crab – HP		Invertebrates and Shellfish				
	English Sole (n = 21)	Pacific Staghorn Sculpin (n = 24)	Shiner Surfperch (n = 27)	Starry Flounder (n = 3)	English Sole – with Skin (n = 7)	English Sole – without Skin (n = 6)	Pile Perch – with Skin (n = 1)	Starry Flounder with Skin (n = 1)	Striped Perch – with Skin (n = 1)	Dungeness Crab (n = 9)	Slender Crab (n = 12)	Dungeness Crab (n = 4)	Slender Crab (n = 4)	Amphipod – WB (n = 4)	Benthic Invert – WB (n = 20)	Mussels, Wild – EM (n = 22)	Mussels, Transplanted – EM (n = 34)	Soft-shell Clam – WB (n = 14)
PAHs																		
Total HPAH (calc'd)	100	4	89	100	57	0	100	100	100	78	100	50	100	25	100	95	71	100
Total LPAH (calc'd)	100	8	93	100	100	0	100	100	100	78	100	75	100	0	100	0	29	100
Total PAH (calc'd)	100	12	93	100	100	0	100	100	100	78	100	75	100	25	100	95	71	100
Phthalates[a]																		
BBP	14	0	29	0	0	0	0	0	0	0	0	25	75	0	5	0	0	0
Dimethyl phthalate	0	0	0	0	0	0	0	0	0	0	8	0	25	0	0	0	0	0
Chlorobenzenes																		
Hexachlorobenzene	19	8	4	0	14	0	0	0	0	11	0	25	0	0	0	0	0	64
Other SVOCs																		
2,4-Dimethylphenol	0	4	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0
4-Methylphenol	0	17	4	0	0	0	0	0	0	0	0	0	0	0	30	0	6	50
Benzoic acid	67	96	63	100	86	0	100	100	0	0	0	0	0	0	95	100	100	100
Benzyl alcohol	52	29	22	0	0	0	100	0	0	0	8	25	0	0	40	5	15	0
Carbazole	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
n-Nitrosodiphenylamine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0
Pentachlorophenol	24	12	7	33	0	0	0	0	0	0	0	0	0	0	25	0	0	0
Phenol	0	8	48	0	0	0	0	0	0	0	8	0	0	50	45	0	0	86

[a] BEHP data are summarized in Table 4-63 in Section 4.2.6.4.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
EM – edible meat
HP – hepatopancreas
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound

4.2.9.4 Surface water

On multiple occasions, surface water samples were collected in the LDW near three CSO outfalls and analyzed for SVOCs as part of the King County WQA (King County 1999e). Three SVOCs were detected in surface water samples: benzoic acid, di-n-butyl phthalate, and phenol (Table 4-82).[100] Detection frequencies of these three chemicals were generally low, ranging from 2 to 38% at the three locations. Di-n-butyl phthalate was detected at all three locations, with concentrations ranging from 0.25 to 1.2 µg/L. However, it should be noted that phthalates were often detected in the method blank samples, and, therefore, these results could be influenced by laboratory contamination. During data validation, detected sample concentrations were compared with the detected method blank concentrations, and sample concentrations less than 10 times the concentration in the associated method blank samples were qualified as undetected at elevated RLs (nine results for di-n-butyl phthalate). Benzoic acid was detected at the two downstream locations, with concentrations ranging from 0.98 to 1.5 µg/L. Phenol was detected once, at RM 1.9, at a concentration of 2.01 µg/L. Concentrations of SVOCs in surface water samples are compared with WQC in Section 4.2.2 (Table 4-20).

Table 4-82. Summary of surface water data for detected SVOCs at three locations in the LDW

CHEMICAL BY LOCATION ID[a, b]	DETECTION FREQUENCY RATIO	%	CONCENTRATION (µg/L) MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLs[c]
BRN (RM 1.1)					
Benzoic acid	5/39	13	0.98	1.4	0.94 – 0.97
Di-n-butyl phthalate	2/39	5	0.25	0.26	0.24
SWM (RM 1.9)					
Benzoic acid	3/42	7	1.1	1.5	0.94 – 0.97
Di-n-butyl phthalate	2/42	5	0.27	0.483	0.24
Phenol	1/42	2	2.01	2.01	0.94 – 0.97
NFK (RM 4.9)					
Di-n-butyl phthalate	5/13	38	0.39	1.2	0.24 – 0.46
Phenanthrene	0/13	0	nd	nd	0.14 – 0.17

[a] BRN, SWM, and NFK are located in the LDW near the Brandon, SW Michigan, and Norfolk CSOs, respectively.

[b] BEHP data are summarized in Table 4-64 in Section 4.2.6.5.

[c] RLs are based only on non-detect samples.

BEHP – bis(2-ethylhexyl) phthalate
ID – identification
RL – reporting limit
RM – river mile
SVOC – semivolatile organic compound

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[100] BEHP data are presented in Section 4.2.6.5.

PAHs were not detected in any of the surface water samples collected by King County during the WQA (King County 1999e).

Because many organic compounds are not typically detected in ambient grab samples, particularly in the dissolved phase, King County also deployed SPMDs to collect data for selected organic compounds, including PAHs, at 1 and 3 m below the water surface near the Duwamish/Diagonal (RM 0.5) and Brandon (RM 1.1) CSO outfalls, as described in Section 4.1.2.4. Ambient PAH concentrations in water were calculated using the mass of chemicals detected in each SPMD and the chemical-specific partition coefficients between the polyethylene and water, which were derived from laboratory experiments conducted at Battelle's Marine Science Laboratory.

Calculated concentrations of individual PAHs ranged from 0.0000027 to 0.030 µg/L (Table 4-83). None of the calculated water concentrations for PAHs were higher than the lowest RLs achieved for PAHs during the analysis of the grab water samples (0.100 µg/L). The individual PAHs with the highest calculated concentrations were acenaphthene (0.030 µg/L), phenanthrene (0.025 µg/L), fluorene (0.020 µg/L), and fluoranthene (0.020 µg/L).

Table 4-83. PAH concentrations in LDW surface water calculated from SPMDs deployed near the Duwamish/Diagonal and Brandon CSOs

	CALCULATED CONCENTRATION (µg/L)			
	DUWAMISH/DIAGONAL CSO		BRANDON CSO	
CHEMICAL	1-m Depth	3-m Depth	1-m Depth	3-m Depth
Acenaphthene	0.0300	0.016	0.019	0.018
Acenaphthylene	0.0016	0.00091	0.0008	0.00071
Anthracene	0.0023	0.0016	0.0016	0.0014
Benzo(a)anthracene	0.00036	0.00021	0.00022	0.00017
Benzo(a)pyrene	0.000033	0.00003	0.000024	0.00002
Benzo(b)fluoranthene	0.00015	0.00011	0.000095	0.000081
Benzo(g,h,i)perylene	0.000048	0.000061	0.00003	0.000027
Benzo(k)fluoranthene	0.00012	0.000027	0.000078	0.000066
Chrysene	0.0004	0.00024	0.00021	0.00017
Dibenzo(a,h)anthracene	0.0000044	0.0000052	0.0000034	0.0000027
Fluoranthene	0.02	0.007	0.0075	0.0063
Fluorene	0.02	0.011	0.011	0.0099
Indeno(1,2,3-c,d)pyrene	0.000022	0.000019	0.000015	0.0000093
Naphthalene	0.0016[a]	nd	0.00066[a]	nd
Phenanthrene	0.025	0.013	0.013	0.011
Pyrene	0.007	0.0038	0.0038	0.0031

Note: Unvalidated PAH data are available for four sampling events conducted by King County in 2003 and 2004 at one sampling location at RM 0.2 (Williston 2009). Surface water grab samples were collected at 1 m below the surface and 1 m above the bottom during these events. Of the 17 PAH compounds analyzed, six were detected (acenaphthene, fluoranthene, fluorene, naphthalene, phenanthrene, and pyrene) at concentrations ranging from 0.0098 to 0.0842 µg/L. Method detection limits ranged from 0.0094 to 0.095 µg/L.

[a] Results for naphthalene are highly uncertain because high concentrations were found in the trip blanks (Mickelson 2007).

CSO – combined sewer overflow

LDW – Lower Duwamish Waterway

nd –non-detect

SPMD – semi-permeable membrane device

4.2.9.5 Seeps

SVOCs were analyzed in 3 seep water samples collected at T-117; 7 seep water samples collected at Rhône-Poulenc in 1995; 9 and 5 seep water samples collected at GWI in 1998 and 1999, respectively; 17 seep water samples collected at Boeing Plant 2; and 16 seep water samples collected throughout the LDW as part of the RI in 2004 (Map 4-11a). Detected concentrations of SVOCs in seep water samples are shown on Maps 4-11b through 4-11e.

SVOCs were detected in one unfiltered seep water sample from the Boeing Plant 2 sampling eventand in unfiltered and filtered seep water samples collected at one location (SP-54 in EAA 2) during the RI event (Table 4-84).[101] SVOCs were also detected in seep water samples collected at GWI; it is not known if these samples were filtered. SVOCs concentrations in seep water samples are compared with WQC in Section 4.2.2 (Table 4-21).

Table 4-84. Summary of seep water data for detected SVOCs

Sampling Event/Year	Chemical[a, b]	Detection Frequency		Detected Concentration (µg/L)	RL (µg/L)[c]
		Ratio	%		
Boeing Plant 2 (1995)	fluoranthene	1/17	6	1.3	1.0
	phenanthrene	1/17	6	1.0	1.0
GWI (1998)	1,2-dichlorobenzene	1/9	11	1.3	1.0
GWI (1999)	acenaphthene	1/5	20	1.3	0.5
	1,2-dichlorobenzene	1/5	20	1.0	0.5
LDW RI (2004)	**Unfiltered**				
	1,2-dichlorobenzene	1/16	6	2.9	1.0
	1,3-dichlorobenzene	1/16	6	58.3	1.0
	1,4-dichlorobenzene	1/16	6	40.2	1.0
	Filtered				
	1,3-dichlorobenzene	1/15	7	3.6	1.0
	1,4-dichlorobenzene	1/15	7	3.9	1.0

[a] BEHP data are summarized in Table 4-65 in Section 4.2.6.6.

[b] Samples collected at Boeing Plant 2, Rhône-Poulenc, and T-117 were unfiltered, as were some of the samples collected for the RI; it is not known if samples collected from GWI were filtered.

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[101] BEHP data for seep water are presented in Section 4.2.6.6.

[c] RLs are based only on non-detect samples.

BEHP – bis(2-ethylhexyl) phthalate	RI – remedial investigation
GWI – Great Western International	RL – reporting limit
J – estimated concentration	SVOC – semivolatile organic compound
nr – not reported	T-117 – Terminal 117

4.2.9.6 Porewater

The only porewater samples from the LDW that have been analyzed for SVOCs were those collected during the 2004 Rhône-Poulenc sampling event. Table 4-85 summarizes concentrations of SVOCs that were detected in at least one porewater sample from this event.[102] Five individual PAHs were detected in the sample from location SH-02 at concentrations up to 0.32 µg/L (Map 4-12d). SVOC concentrations in porewater samples are compared with WQC in Section 4.2.2 (Table 4-22).

Table 4-85. Summary of porewater data from Rhône-Poulenc for detected SVOCs

Chemical[a]	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
Benzo(a)pyrene	1/9	11	0.0320 J	0.0320 J	0.37
Benzo(b)fluoranthene	1/9	11	0.0390 J	0.0390 J	0.37
Benzo(k)fluoranthene	1/9	11	0.0390 J	0.0390 J	0.37
Fluoranthene	1/9	11	0.26 J	0.26 J	0.37
Pyrene	1/9	11	0.32 J	0.32 J	0.37

[a] BEHP data are summarized in Table 4-65 in Section 4.2.6.7.

[b] RLs are based only on non-detect samples.

BEHP – bis(2-ethylhexyl) phthalate
J – estimated concentration
PAH – polycyclic aromatic hydrocarbon
RL – reporting limit
SVOC – semivolatile organic compound

4.2.9.7 Summary of SVOC data

PAHs and BEHP were the most frequently detected SVOCs in surface sediment.[103] The mean concentrations of total HPAHs and total LPAHs were 4,000 and 700 µg/kg dw, respectively. Areas with the highest concentrations of PAHs (i.e., greater than the numerical 95th percentile) were generally located in the following areas: the Duwamish/Diagonal EAA, Slip 1, between RM 1.3 and RM 1.5 on the west side, Slip 4, the Boeing Plant 2/Jorgensen Forge EAA, T-117, RM 3.8 EAA, and Slip 6.

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[102] BEHP data for porewater are presented in Section 4.2.6.7.

[103] BEHP data are summarized in Section 4.2.6.

At least one PAH compound was detected in nearly all surface sediment samples, and PAHs were also detected frequently in tissue. In subsurface sediment, areas with the highest total LPAH and HPAH concentrations included RM 0.5 on the west side; RM 1.2 on the east side; RM 1.4 on the west side; RM 2.7 on the west side; Slips 1, 3, 4, and 6; the Duwamish/Diagonal EAA; and the Boeing Plant 2/Jorgensen Forge EAA.

In general, individual PAHs were detected frequently in most tissue types. Although low-level analytical methods were used for all tissue samples collected in 2004, RLs were high for Pacific staghorn sculpin samples because analytical dilutions were used to minimize matrix interferences, resulting in reduced sensitivity of the analysis. PAHs were infrequently detected in Pacific staghorn sculpin at these elevated RLs. Benzoic acid, which was the second most frequently detected SVOC in tissue, was detected frequently in fish, benthic invertebrates, mussels, and clams, but was not detected in any crab samples. Other SVOCs, most notably the phthalates that were commonly detected in sediments, were generally infrequently detected in tissue, even when highly sensitive analytical methods were applied.

SVOCs were detected infrequently in surface water, seep, and porewater samples collected from the LDW.

4.2.10 Organochlorine pesticides

This section summarizes the nature and extent of organochlorine pesticide concentrations in surface sediment, subsurface sediment, tissue, surface water, and seep water. None of the porewater samples collected from the LDW were analyzed for organochlorine pesticides.

Some of the pesticide data are uncertain because of analytical interference from the presence of PCB congeners, as identified by both the analytical laboratory and the data validators during the pesticide analyses of benthic invertebrate, fish, and crab tissue as part of the RI. Samples with this probable interference were assigned a JN-qualifier, indicating the results are highly uncertain and biased high. A more detailed discussion of this data quality issue is provided in Section 4.1.3.2.

4.2.10.1 Surface sediment

Table 4-86 presents a summary of organochlorine pesticides detected in surface sediment. Most organochlorine pesticides were detected in 6% or fewer of the surface sediment samples in which they were analyzed. 4,4′-DDD (dichlorodiphenyldichloroethane), 2,4′-DDT, 4,4′-DDT, and gamma-chlordane were the organochloride pesticides most frequently detected in surface sediment, in 34, 31, 21, and 17% of the samples in which they were analyzed, respectively. Other DDT compounds were detected at frequencies of 5 to 31%, and other chlordane compounds were detected at frequencies of 9 to 11%.

Table 4-86. Summary of surface sediment data for detected organochlorine pesticides

Pesticide	Detection Frequency		Concentration (µg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
DDT and Metabolites					
2,4′-DDD	5/93	5	1.6 JN	10 JN	0.97 – 34
2,4′-DDE	2/93	2	2.8 JN	11 JN	0.97 – 34
2,4′-DDT	29/93	31	0.24 JN	11 JN	1.9 – 460
4,4′-DDD	66/193	34	0.29 JN	840	0.81 – 540
4,4′-DDE	29/193	15	0.28 JN	370 J	0.81 – 800
4,4′-DDT	40/193	21	0.48 JN	1,700	0.81 – 56
Total DDTs	76/193	39	0.72 JN	2,900 J	nc
Chlordane Compounds					
alpha-Chlordane	13/149	9	0.10 JN	36	0.72 – 30
gamma-Chlordane	25/149	17	0.20 JN	200	0.72 – 96
Chlordane	5/44	11	25	62	7.7 – 330
Total chlordanes	28/149	19	0.20 JN	230	nc
Aldrin/Dieldrin					
Aldrin	4/193	2	0.014 J	1.6 JN	0.40 – 56
Dieldrin	8/193	4	0.099 J	280	0.81 – 91
Other Pesticides					
alpha-BHC	3/193	2	0.14 J	1.8 JN	0.40 – 56
beta-BHC	4/193	2	0.087 J	13	0.40 – 56
gamma-BHC	11/193	6	0.050 J	6.7 JN	0.40 – 56
delta-BHC	3/155	2	0.081 J	11 JN	0.40 – 56
alpha-Endosulfan	9/147	6	0.18 JN	71 JN	0.40 – 100
beta-Endosulfan	3/149	2	0.47 J	10 JN	0.81 – 200
Endosulfan	1/46	2	0.11 J	0.11 J	0.81 – 56
Endosulfan sulfate	3/191	2	0.63 JN	25 JN	0.81 – 200
Endrin	4/193	2	0.99 JN	9.1	0.81 – 200
Endrin aldehyde	6/182	3	0.28 JN	130	0.81 – 250
Endrin ketone	7/137	5	0.83 JN	110 JN	0.81 – 200
Heptachlor	6/193	3	0.12 J	5.2	0.43 – 70
Heptachlor epoxide	5/193	3	0.47 JN	4.9 JN	0.40 – 510
Methoxychlor	11/193	6	0.34 JN	99	0.97 – 330
Mirex	3/93	3	0.29 JN	1.0 JN	0.97 – 34
Toxaphene	2/191	1	340 JN	6,300 JN	1.0 – 4,300

[a] RLs based only on non-detect samples.

BHC – benzene hexachloride	J – estimated concentration
DDD – dichlorodiphenyldichloroethane	LDW – Lower Duwamish Waterway
DDE – dichlorodiphenyldichloroethylene	N – tentative identification
DDT – dichlorodiphenyltrichloroethane	nc – not calculated
dw – dry weight	RL – reporting limit

The highest concentration of total DDTs (2,900 µg/kg ww) in surface sediment was detected in sample DR178 collected in Slip 4 (Map 4-68). This location also had a high PCB concentration in surface sediment (7,000 µg/kg dw), indicating a potential for analytical interference. Other areas with relatively high total DDT concentrations in surface sediment were located between RM 0.4 and RM 0.6 on the east side of the LDW (Duwamish/Diagonal EAA), the inlet at RM 2.2 on the west side of the LDW (EAA 2), and at the mouth of Slip 3.

4.2.10.2 Subsurface sediment

Organochlorine pesticides were analyzed in a subset of subsurface sediment samples collected in the LDW. Of the samples analyzed for organochlorine pesticides, 17% had one or more detected DDT compounds (Table 4-87). Other organochlorine pesticides were detected in 17% or fewer of the samples in which they were analyzed.

Table 4-87. Summary of subsurface sediment data for organochlorine pesticides from any interval

	Detection Frequency		Concentration (µg/kg dw)[a]		
Pesticide	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
DDT and Metabolites					
2,4′-DDD	0/40	0	nd	nd	1.9 – 29
2,4′-DDE	0/40	0	nd	nd	1.9 – 100
2,4′-DDT	0/40	0	nd	nd	1.9 – 29
4,4′-DDD	13/95	14	0.27 J	10	1.5 – 39
4,4′-DDE	14/95	15	0.45 J	18 J	0.62 – 90
4,4′-DDT	5/95	5	0.64 J	2.8	0.96 – 160
Total DDTs	16/95	17	0.27 J	20 J	nc
Chlordane Compounds					
alpha-Chlordane	1/89	1	0.44 J	0.44 J	0.052 – 14
gamma-Chlordane	4/87	5	0.22 J	0.71 J	0.92 – 82
Chlordane	1/6	17	29.0	29.0	7.5 – 35
Total chlordanes	4/89	4	0.22 J	1.59 J	nc
Aldrin/Dieldrin					
Aldrin	1/95	1	1.5 J	1.5 J	0.33 – 16.00
Dieldrin	3/95	3	0.80 J	1.8	0.14 – 95.00

PESTICIDE	DETECTION FREQUENCY		CONCENTRATION (µg/kg dw)[a]		
	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLs[b]
Other Pesticides					
alpha-BHC	3/71	4	0.20 J	0.47 J	0.086 – 14
beta-BHC	0/66	0	nd	nd	0.96 – 24
gamma-BHC	0/95	0	nd	nd	0.099 – 14
delta-BHC	7/48	15	7.0	1,100	0.96 – 14
alpha-Endosulfan	0/62	0	nd	nd	0.96 – 14
beta-Endosulfan	0/62	0	nd	nd	1.90 – 29
Endosulfan	0/4	0	nd	nd	1.5 – 1.6
Endosulfan sulfate	0/62	0	nd	nd	1.90 – 69
Endrin	0/66	0	nd	nd	1.5 – 150
Endrin aldehyde	1/66	2	10	10	1.5 – 29
Endrin ketone	0/62	0	nd	nd	1.90 – 29
Heptachlor	2/95	2	0.25 J	1.3	0.097 – 14
Heptachlor epoxide	0/66	0	nd	nd	0.96 – 120
Methoxychlor	0/66	0	nd	nd	1.0 – 140
Mirex	0/40	0	nd	nd	1.9 – 29
Toxaphene	0/66	0	nd	nd	10 – 1,400

Note: Data included in the subsurface dataset are described in Section 4.1.2.2; subsurface sediment samples collected from areas that were subsequently dredged are not summarized in this table because they do not represent existing conditions.

[a] Mean concentrations were not calculated because sampling was generally focused in contaminated areas, resulting in a non-random sampling pattern, which is not likely representative of the entire LDW.

[b] RLs are based only on non-detect samples.

BHC – benzene hexachloride
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
dw – dry weight
J – estimated concentration
LDW – Lower Duwamish Waterway
N – tentative identification
nc – not calculated
nd – not detected at any location
RL – reporting limit

Total DDT concentrations in subsurface sediment ranged from 0.27 to 20 µg/kg dw, with the three highest concentrations (17, 18, and 20 µg/kg dw) detected during the EPA SI sampling conducted in 1998. These concentrations were detected in the 2-to-4-ft interval of core DR025 at RM 1.2, in the 0-to-2-ft interval of core DR137 at RM 2.2 near EAA 2, and in the 2-to-4-ft interval of core DR021 at RM 0.9 at the mouth of Slip 1, respectively.

4.2.10.3 Tissue

Table 4-88 presents the detection frequency and range of organochlorine pesticide concentrations in LDW tissue samples. 2-4′-DDT and gamma chlordane were the most frequently detected pesticides (91 and 85%, respectively, of all tissue samples in which they were analyzed). Other DDT compounds were detected at a frequency of 8 to 78%, and other chlordane compounds were detected at a frequency of 0 to 33% in all tissue samples in which they were analyzed. Heptachlor epoxide, alpha-endosulfan, and beta-BHC were detected in 33, 26, and 26%, respectively, of all tissue samples in which they were analyzed. The remaining organochlorine pesticides were detected in 16% or fewer of all tissue samples in which they were analyzed. The maximum tissue concentrations of most organochlorine pesticides have a JN-qualifier, as shown in Table 4-88. This qualifier indicates probable interference in the analysis from PCB congeners, resulting in data that are highly uncertain and biased high.

Table 4-88. Summary of organochlorine pesticide data in all tissue samples

	Detection Frequency		Concentration (µg/kg ww)		
Chemical	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
DDT and Metabolites					
2,4′-DDD	46/179	26	0.49 J	57 JN	0.18 – 88
2,4′-DDE	14/179	8	0.72 J	110 JN	0.26 – 45
2,4′-DDT	163/179	91	1.8 J	440 JN	0.22 – 34
4,4′-DDD	126/212	59	0.23 JN	20 JN	0.13 – 18
4,4′-DDE	155/212	73	0.70 JN	46 JN	1.0 – 14
4,4′-DDT	165/212	78	0.56 J	470 JN	0.38 – 82
Total DDTs	186/212	88	1.1	1,020 JN	nc
Chlordane Compounds					
alpha-Chlordane	44/188	23	0.60 JN	12 JN	0.36 – 18
gamma-Chlordane	160/188	85	0.52 J	330 JN	0.14 – 56
Oxychlordane	6/18	33	0.23 J	1.5 J	0.12 – 1.0
cis-Nonachlor	0/18	0	nd	nd	0.12 – 2.3
trans-Nonachlor	6/18	33	0.70 J	3.2	0.14 – 1.5
Chlordane	0/24	0	nd	nd	6.7
Total chlordane	165/188	88	0.24 J	330 JN	nc
Aldrin/Dieldrin					
Aldrin	6/212	3	0.61 J	6.2 JN	0.20 – 18
Dieldrin	12/212	6	0.76 J	5.7	0.11 – 34
Other Pesticides					
alpha-BHC	13/212	6	0.20 JN	21 JN	0.16 – 18
alpha-Endosulfan	55/212	26	0.21 J	30 JN	0.13 – 15
beta-BHC	55/212	26	0.27 JN	15 JN	0.38 – 18
beta-Endosulfan	33/212	16	0.94 J	44 JN	0.35 – 18
delta-BHC	3/212	1	0.51 JN	2.2 JN	0.34 – 18
Endosulfan sulfate	4/212	2	0.73 J	2.3	0.27 – 190

Chemical	Detection Frequency		Concentration (µg/kg ww)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Endrin	34/212	16	0.10 JN	40 JN	0.099 – 72
Endrin aldehyde	31/212	15	0.42 JN	78 JN	0.17 – 18
Endrin ketone	4/179	2	0.60 JN	8.4 JN	0.29 – 18
gamma-BHC	17/212	8	0.51 JN	7 JNM	0.28 – 18
Heptachlor	10/212	5	0.96 J	9.7 JN	0.45 – 18
Heptachlor epoxide	69/212	33	0.72 J	45 JN	0.15 – 18
Methoxychlor	11/212	5	0.49	130 JN	0.27 – 23
Mirex	0/179	0	nd	nd	0.27 – 18
Toxaphene	0/212	0	nd	nd	10 – 4,800

[a] RLs are based only on non-detect samples.

BHC – benzene hexachloride
DDD – dichlorodiphenyldichloroethane
DDE - dichlorodiphenyldichloroethylene
DDT - dichlorodiphenyltrichloroethane
J – estimated concentration
M – calculated whole-body concentration
N – tentative identification
nc – not calculated
nd – not detected
RL – reporting limit
ww – wet weight

4.2.10.4 Surface water

Surface water grab samples collected during the pilot study for the King County WQA (King County 1999e) did not contain detectable concentrations of organochlorine pesticides. Instead, organochlorine pesticides were analyzed in the SPMDs deployed at 1 and 3 m below the water surface just offshore of the Duwamish/Diagonal and Brandon CSOs (at RM 0.5 and RM 1.1, respectively), as described in Section 4.1.2.4. Organochlorine pesticide concentrations in the SPMDs were used to calculate concentrations in surface water using the mass of chemicals detected in each SPMD and the chemical-specific partition coefficients between polyethylene and water, which were derived from laboratory experiments conducted at Battelle's Marine Science Laboratory. Calculated concentrations were very low, ranging from 0.000000034 to 0.000033 µg/L (Table 4-89). Calculated pesticide concentrations in surface water samples are compared with WQC in Section 4.2.2 (Table 4-20).

Table 4-89. Organochlorine pesticide concentrations in surface water calculated from SPMDs deployed near Duwamish/Diagonal and Brandon CSOs

Pesticide	Calculated Concentration (µg/L)			
	Duwamish/Diagonal CSO		Brandon CSO	
	1-m Depth	3-m Depth	1-m Depth	3-m Depth
2,4′-DDD	0.000013	0.0000067	0.0000079	0.0000063
4,4′-DDD	0.000015	0.0000091	0.000010	0.0000085
4,4′-DDE	0.0000065	0.0000019	0.00000093	0.000000034

PESTICIDE	CALCULATED CONCENTRATION (µg/L)			
	DUWAMISH/DIAGONAL CSO		BRANDON CSO	
	1-m Depth	3-m Depth	1-m Depth	3-m Depth
4,4′-DDT	nd	nd	nd	nd
Aldrin	0.0000059	0.0000040	0.0000037	0.0000030
alpha-Chlordane	0.0000026	0.0000017	0.0000015	0.0000013
gamma-Chlordane	0.0000013	0.000000077	0.00000011	nd
Dieldrin	0.000033	0.000023	nd	0.000020
Endrin	0.000014	0.0000089	0.0000064	nd

Note: Unvalidated pesticide data are available for four sampling events conducted in 2003 and 2004 by King County at one sampling location at RM 0.2 (Williston 2009). Surface water grab samples were collected at 1 m below the surface and 1 m above the bottom during these events. Pesticides were not detected at method detection limits ranging from 0.0047 to 0.056 µg/L.

[a] Value for chlordane.

DDD – dichlorodiphenyldichloroethane
DDE - dichlorodiphenyldichloroethylene
DDT - dichlorodiphenyltrichloroethane
CSO – combined sewer overflow
nd – not detected
SPMD – semi-permeable membrane device

4.2.10.5 Seep water

The only LDW seep water samples analyzed for organochlorine pesticides were those collected as part of the RI (Windward 2004b). Heptachlor epoxide (a degradation product of heptachlor) was detected in both the filtered and unfiltered seep water samples collected at Seep 39, at concentrations of 0.0090 and 0.0076 µg/L, respectively (Map 4-11e). No other organochlorine pesticides were detected in seep water samples. Pesticide concentrations in seep water samples are compared with WQC in Section 4.2.2 (Table 4-21).

4.2.10.6 Summary

Organochlorine pesticides were analyzed in a subset of surface sediment, subsurface sediment, tissue, and seep water samples. Analytical interference in the organochlorine pesticide analyses from the presence of PCBs in the samples resulted in high uncertainty and high bias in some of the organochlorine pesticide data, particularly samples with high PCB concentrations.

4,4′-DDD, 2,4′-DDT, 4,4′-DDT, and gamma-chlordane were the organochlorine pesticides most frequently detected in surface sediment, in 34, 31, 21, and 17% of the samples in which they were analyzed, respectively. The highest total DDT concentration in surface sediment (2,900 µg/kg dw) was detected in a sample collected from Slip 4; this sample also had a relatively high total PCB concentration (7,000 µg/kg dw). Organochlorine pesticides were infrequently detected in subsurface sediment; 4,4′-DDE (dichlorodiphenyldichloroethylene) was most frequently detected (in 15% of the samples in which it was analyzed), at a maximum concentration of 18 µg/kg dw.

DDT and chlordane compounds were frequently detected in tissue samples, but there is high uncertainty in these data because of analytical interference by PCBs. Organochlorine pesticide concentrations in surface water were estimated from SPMDs deployed at two locations in the LDW. Calculated concentrations of organochlorine pesticides in surface water ranged from 0.000000034 to 0.000033 µg/L. Organochlorine pesticides were detected in one seep sample collected as part of the RI (Seep 39 near RM 4.3 on the western shoreline); heptachlor epoxide was detected in both the filtered and unfiltered samples from this location at concentrations of 0.0090 and 0.0076 µg/L, respectively.

4.2.11 VOCs

This section summarizes the nature and extent of VOC concentrations in surface sediment, subsurface sediment, seep water, and porewater samples collected from the LDW.

4.2.11.1 Surface sediment

VOCs were analyzed in 36 surface sediment samples collected throughout the LDW as part of the 1998 EPA SI, in 3 surface sediment samples collected at RM 3.3 as part of the 1995 Boeing Plant 2 RFI, and in 2 surface sediment samples collected near RM 0.5 as part of the 1991 Harbor Island RFI.

The highest VOC concentration (dichloromethane at 1,600 µg/kg dw) was detected in sample DR008 collected near RM 0.5 (Table 4-90). The second highest VOC concentration (acetone at 1,000 µg/kg dw) was detected in sample DR047 collected near RM 0.9 south of Kellogg Island. The remaining detected VOC concentrations were less than 160 µg/kg dw.

Table 4-90. Summary of surface sediment data for detected VOCs

	Detection Frequency		Concentration (µg/kg dw)		
Chemical	Ratio	%	Minimum Detect	Maximum Detect	Range of RLs[a]
Acetone	3/41	7	110 J	1,000 J	11 – 21,000
Carbon disulfide	12/41	29	0.84 J	4.0 J	1.4 – 1,100
p-Cymene	3/36	8	1.6 J	25	1.5 – 530
Dichloromethane	1/41	2	1,600	1,600	2.8 – 21
Methyl ethyl ketone	15/41	37	5.3	35	3.0 – 1,100
Tetrachloroethene	2/41	5	0.21 J	0.52 J	1.4 – 530
Toluene	3/41	7	1.0 J	2.2 J	1.4 – 530

[a] RLs based only on non-detect samples.

dw – dry weight

J – estimated concentration

RL – reporting limit

VOC – volatile organic compounds

4.2.11.2 Subsurface sediment

VOCs were analyzed in subsurface core samples collected from eight locations between RM 3.3 and RM 3.5 as part of four sampling events. As part of the RI, one subsurface core was analyzed for VOCs because of elevated readings from the photoionization detector during core processing. VOCs were not analyzed in other cores because the detector did not indicate that they were present at elevated concentrations. This core was collected at LDW-SC49 between RM 3.5 and RM 3.6 and analyzed at 1-ft intervals to a depth of 12 ft below the sediment surface.

Detected VOC concentrations in subsurface sediment ranged from 1.5 to 200,000 µg/kg dw (Table 4-91). The most frequently detected VOCs were acetone, carbon disulfide, and methyl ethyl ketone, with detection frequencies of 81, 50, and 44% of the samples in which they were analyzed, respectively. With the exception of these chemicals, individual VOCs were detected in less than 25% of samples.

Table 4-91. Summary of subsurface sediment data for detected VOCs

	Detection Frequency		Concentration (µg/kg dw)[a]		
Chemical	Ratio	%	Minimum Detect	Maximum Detect	Range of RLs[b]
1,1-Dichloroethene	2/32	6	120	320	0.90 – 9.6
1,2,4-Trimethylbenzene	3/24	13	14 J	580 J	0.90 – 39
1,3,5-Trimethylbenzene	4/24	17	1.9	350 J	0.90 – 39
Acetone	26/32	81	8.20	640	6.6 – 37
Benzene	5/32	16	4.4	62	0.90 – 9.6
Carbon disulfide	16/32	50	1.30	18	0.90 – 9.6
Chloroform	1/32	3	1.8	1.8	0.90 – 9.6
cis-1,2-Dichloroethene	7/32	22	2.2	200,000	0.90 – 9.6
p-Cymene	5/24	21	4.1 J	130 J	0.90 – 39
Dichloromethane	10/32	31	1.5 J	5.90	1.90 – 11
Ethylbenzene	3/58	5	27	360 J	0.75 – 9.6
Isopropylbenzene	3/24	13	3.9 J	87 J	0.90 – 39
Methyl ethyl ketone	14/32	44	9.60	100	4.60 – 28
n-Butylbenzene	2/24	8	58	78 J	0.90 – 39
n-Propylbenzene	3/24	13	1.8 J	100 J	0.90 – 39
sec-Butylbenzene	2/24	8	57	57	0.90 – 39
tert-Butylbenzene	1/24	4	6.1	6.1	0.90 – 39
Toluene	7/32	22	1.8	8,300	0.90 – 9.6
trans-1,2-Dichloroethene	3/32	9	22	1,700 J	0.90 – 9.6
Trichloroethene	4/58	7	3.6 J	23	0.37 – 9.6
Vinyl chloride	5/32	16	29	60,000	0.90 – 11
Xylene (ortho)	3/52	6	31	610 J	0.90 – 9.6
Xylene (meta and para)	3/41	7	74 J	4,700	1.1 – 9.6
Total xylenes (calc'd)	3/52	6	105 J	5,300 J	nc

Note: Data included in the subsurface dataset are described in Section 4.1.2.2; subsurface sediment samples collected from areas that were subsequently dredged are not summarized in this table because they do not represent existing conditions.

[a] Mean concentrations were not calculated because sampling was generally focused in contaminated areas, resulting in a non-random sampling pattern, which is not likely representative of the entire LDW.

[b] RLs based only on non-detect samples.

dw – dry weight

J – estimated concentration

LDW – Lower Duwamish Waterway

RL – reporting limit

VOC – volatile organic compounds

The highest concentrations of VOCs were detected in the 9-to-10-ft, 10-to-11-ft, and 11-to-12-ft intervals of core LDW-SC49b, collected in the navigation channel at RM 3.55. The VOCs with the highest concentrations in this core were cis-1,2-dichloroethene (200,000 and 140,000 µg/kg dw in the 10-to-11-ft and 11-to-12-ft intervals, respectively) and vinyl chloride (60,000 µg/kg dw in the 11-to-12-ft interval). These two VOCs were not detected in samples collected from the top 6 ft of the core.

4.2.11.3 Seep water

VOCs have been analyzed in seep water samples collected at three sites: Boeing Plant 2 (in 1995), Rhône-Poulenc (in 1995), and GWI (during 10 sampling events from 1994 to 1999). In addition, 16 seep water samples collected from throughout the LDW were analyzed for VOCs as part of the RI (Map 4-11a). Table 4-92 and Maps 4-11b through 4-11e present data for detected VOCs in seep water samples from the LDW investigations listed above. No VOCs were detected in seep water samples collected from Rhône-Poulenc. VOC concentrations in seep water samples are compared with WQC in Section 4.2.2 (Table 4-21).

Table 4-92. Concentrations of VOCs with at least one detected value in seep water samples

Chemical	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Boeing Plant 2 – 1995					
Benzene	1/18	6	2.2	2.2	1
cis-1,2-Dichloroethene	6/18	33	1.2	40	1
Trichloroethene	3/18	17	1.9	13	1
Vinyl chloride	4/18	22	3.1	36	0.01 – 2
GWI – April 1994					
1,1-Dichloroethane	3/6	50	1.0	2.0	1.0
1,1-Dichloroethene	1/6	17	1.0	1.0	1.0
Total 1,2-dichloroethene	3/6	50	7.0	110	1.0
Tetrachloroethene	2/6	33	3.0	760	1.0
Trichloroethene	3/6	50	1.0	370	1.0
Vinyl chloride	1/6	17	1.0	1.0	1.0
GWI – July 1994					
1,1-Dichloroethane	1/2	50	1.0	1.0	1.0
1,2-Dichloroethene (total)	2/2	100	3.0	40	na
Tetrachloroethene	2/2	100	2.0	220	na
Trichloroethene	1/2	50	94	94	1.0
Vinyl chloride	1/2	50	3.0	3.0	1.0
GWI – November 1994					
1,2-Dichloroethene (total)	2/7	29	1	50	1.0
Tetrachloroethene	2/7	29	1	440	1.0
Trichloroethene	1/7	14	130	130	1.0
GWI – May 1995					
1,1,1-Trichloroethane	1/7	14	1.0	1.0	1.0
1,1-Dichloroethane	2/7	29	1.0	2.0	1.0
1,2-Dichloroethane	1/7	14	1.0	1.0	1.0
1,2-Dichloroethene (total)	3/7	43	3.0	150	1.0
Tetrachloroethene	2/7	29	6.0	710	1.0
Trichloroethene	2/7	29	8.0	480	1.0
Vinyl chloride	2/7	29	2.0	2.0	1.0
GWI – 1995 annual monitoring					
1,1-Dichloroethane	2/7	29	1.0	1.0	1.0
1,2-Dichloroethane	1/7	14	1.0	1.0	1.0
1,2-Dichloroethene (total)	4/7	57	1.0	160	1.0

Table 4-92, cont. Concentrations of VOCs with at least one detected value in seep water samples

Chemical	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Tetrachloroethene	2/7	29	4.0	610	1.0
Trichloroethene	2/7	29	4.0	360	1.0
Vinyl chloride	2/7	29	1.0	1.0	1.0
GWI – 1996 annual monitoring					
1,1-Dichloroethane	1/5	20	1.0	1.0	1.0
1,2-Dichloroethane	1/5	20	8.0	8.0	1.0
1,2-Dichloroethene (total)	3/5	60	8.0	310	1.0
1,2-Dichloropropane	1/5	20	1.0	1.0	1.0
Tetrachloroethene	2/5	40	13	79	1.0
Toluene	1/5	20	2.	2.0	1.0
Trichloroethene	2/5	40	7.0	58	1.0
Vinyl chloride	1/5	20	3.0 J	3.0 J	1.0
Total xylenes	1/5	20	1.0	1.0	1.0
GWI – 1997 annual monitoring					
1,1-Dichloroethane	1/4	25	1.0	1.0	1.0
1,2-Dichloroethene (total)	2/4	50	37	470	1.0
Tetrachloroethene	2/4	50	19	150	1.0
Trichloroethene	2/4	50	7.0	200	1.0
GWI – 1998 annual monitoring					
1,1-Dichloroethane	2/9	22	4.1	53	1.0
1,1-Dichloroethene	1/9	11	18	18	1.0
1,2-Dichloroethane	1/9	11	8.5	8.5	1.0
1,2-Dichloropropane	1/9	11	2.7	2.7	1.0
Benzene	1/9	11	28	28	1.0
Chlorobenzene	1/9	11	4.1	4.1	1.0
cis-1,2-Dichloroethene	3/9	33	41	3,300	1.0
Tetrachloroethene	4/9	44	1.3	290	1.0
trans-1,2-Dichloroethene	2/9	22	3.5	46	1.0
Trichloroethene	3/9	33	4	170	1.0
Vinyl chloride	2/9	22	1.0	760	1.0
o-Xylene	1/9	11	1.4	1.4	1.0
Total xylenes	1/9	11	1.4	1.4	nc
GWI – 1999 annual monitoring					
1,1-Dichloroethane	1/5	20	58	58	1.0 – 2.0
Benzene	1/5	20	28	28	1.0 – 2.0

Table 4-92, cont. Concentrations of VOCs with at least one detected value in seep water samples

Chemical	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
cis-1,2-Dichloroethene	3/5	60	5.6	3,200	1.0
Tetrachloroethene	2/5	40	3.7	190	1.0 – 20
trans-1,2-Dichloroethene	2/5	40	12	27	1.0
Trichloroethene	2/5	40	2.1	270	1.0 – 20
Vinyl chloride	1/5	20	3,500	3,500	1.0 – 2.0
GWI – 1998 Embayment Study					
1,1-Dichloroethane	5/10	50	1.5	62	1.0
1,1-Dichloroethene	3/10	30	1.0	27	1.0 – 10
1,2,4-Trimethylbenzene	1/4	25	4.1	4.1	10
1,2-Dichloroethane	2/10	20	6.6	27	1.0 – 10
1,2-Dichloropropane	1/10	10	16	16	1.0 – 10
Acetone	3/8	38	1.0	6.4	5.0 – 11
Benzene	2/10	20	21	36	1.0 – 10
Chlorobenzene	1/10	10	8.8	8.8	1.0 – 10
cis-1,2-Dichloroethene	4/4	100	29	5,400	na
Tetrachloroethene	4/10	40	2.6	180	1.0 – 10
Toluene	1/10	10	2.7	2.7	1.0 – 10
trans-1,2-Dichloroethene	3/10	30	16	110	1.0
Trichloroethene	3/10	30	6.0	110	1.0 – 10
Vinyl chloride	4/10	40	3.8	1,600	1.0 – 1.0
o-Xylene	1/10	10	2.8	2.8	1.0 – 10
Total xylenes	1/10	10	2.8	2.8	nc
LDW RI – 2004					
Carbon disulfide	2/16	13	1.1	2.4	1.0
Chlorobenzene	1/16	6	6.5	6.5	1.0

[a] RLs based only on non-detect samples.

GWI – Great Western International
J – estimated concentration
LDW – Lower Duwamish Waterway
na – not applicable
nc – not calculated (RLs not available)
nr – not reported (only detected chemicals reported in the data report; RLs for undetected results not available)
RI – remedial investigation
RL – reporting limit
VOC – volatile organic compound

At Boeing Plant 2, four VOCs were detected in seep water samples. The highest VOC concentration (40 µg/L of cis-1,2-dichloroethene) was detected in seep sample SE-11001.

At GWI, numerous VOCs were detected during the various sampling events. The VOCs with the highest detected concentrations were cis-1,2-dichloroethene (5,400 µg/L in S-13), tetrachloroethene (760 µg/L in S-2), trichloroethene (480 µg/L in S-2), and vinyl chloride (3,500 µg/L in S-13). The seep locations where VOCs were most frequently detected and where the highest concentrations were found were S-1, S-2, S-4, and S-13.

As part of the RI, two VOCs were detected in seep water samples. Carbon disulfide was detected in seep water from two locations (SP- 80 at 1.1 µg/L and SP-54 at 2.4 µg/L), and chlorobenzene was detected in seep water from location SP-54 at a concentration of 6.5 µg/L.

4.2.11.4 Porewater

VOCs were analyzed in porewater samples collected from GWI and Boeing Plant 2/ Jorgensen Forge as part of the RI in 2005 and from Rhône-Poulenc during a sampling event in 2004 (Map 4-12a). VOCs analyzed in the nine porewater samples collected during the Rhône-Poulenc sampling event were benzene, toluene, ethylbenzene, and xylene; none of these VOCs was detected. VOC concentrations in porewater samples are compared with WQC in Section 4.2.2 (Table 4-22).

Table 4-93 and Maps 4-12c and 4-12d present data for detected VOCs in porewater samples collected using peepers in shallow nearshore areas offshore of GWI and Boeing Plant 2/Jorgensen Forge as part of the RI. Porewater samples were also collected from deeper, offshore areas at these two sites using piezometers (see Map 4-12a); VOCs were not detected in any porewater samples collected using piezometers at either site.

Table 4-93. Summary of VOC data for porewater samples collected from locations using peepers

	Detection Frequency[a]		Concentration (µg/L)		
Chemical	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
GWI					
1,1-Dichloroethane	8/10	80	0.4	16	0.2
1,1-Dichloroethene	3/10	30	0.3	4.9	0.2 – 0.4
1,2-Dichlorobenzene	3/10	30	0.5	1.2	0.2 – 0.4
1,2-Dichloroethane	2/10	20	7.4	15	0.2 – 0.4
1,2-Dichloropropane	2/10	20	1.7	2.5	0.2 – 0.4
1,4-Dichlorobenzene	2/10	20	0.3	0.3	0.2 – 0.4
Benzene	1/10	10	9.4	9.4	0.2 – 4.7

CHEMICAL	DETECTION FREQUENCY[a]		CONCENTRATION (µg/L)		
	RATIO	%	MINIMUM DETECT	MAXIMUM DETECT	RL OR RANGE OF RLs[b]
Carbon disulfide	5/10	50	0.2	0.7	0.2 – 0.4
Chlorobenzene	4/10	40	0.3	1.4	0.2 – 0.4
cis-1,2-Dichloroethene	10/10	100	0.5	2,900	na
Isopropylbenzene	2/10	20	0.2	0.3	0.2 – 0.4
Tetrachloroethene	2/10	20	0.4	1.1	0.2 – 0.4
Toluene	5/10	50	0.3	3.5	0.2 – 0.4
trans-1,2-Dichloroethene	7/10	70	0.3	21 J	0.2 – 0.4
Trichloroethene	4/10	40	0.4	2.5	0.2 – 0.4
Vinyl chloride	10/10	100	0.4	2,500	na
Boeing Plant 2/Jorgensen Forge					
1,1-Dichloroethane	1/10	10	0.3	0.3	0.2
cis-1,2-Dichloroethene	7/10	70	0.2	1.7	0.2
Trichloroethene	2/10	20	0.2	0.2	0.2
Vinyl chloride	2/10	20	1.1	13	0.2

[a] Two of the 10 samples from GWI were field replicates collected at PE-08 and two of the 10 samples from Boeing Plant 2 were field replicates collected at PE-10.

[b] RLs are based only on non-detect samples.

GWI – Great Western International
na – not applicable
RL – reporting limit
J – estimated concentration
nd – not detected
VOC – volatile organic compound

The highest concentrations of VOCs were detected in samples collected from GWI. Vinyl chloride and cis-1,2-dichloroethene were detected in all samples collected using peepers and had the highest concentrations. Maximum concentrations of vinyl chloride and cis-1,2-dichloroethene were 2,500 and 2,900 µg/L, respectively, at location PE-06 (Map 4-12c).

At Boeing Plant 2/Jorgensen Forge, four VOCs were detected in porewater samples (Table 4-93). Cis-1,2-dichloroethene was the most frequently detected VOC, in 7 of the 10 samples collected using peepers, at concentrations ranging from 0.2 to 1.7 µg/L. The highest VOC concentration (13 µg/L of vinyl chloride) was detected in a porewater sample collected from location PE-11 (Map 4-12d).

4.2.11.5 Summary

Seven VOCs were detected in surface sediment samples collected from the LDW. The highest concentrations were 1,600 µg/kg dw for dichloromethane and 1,000 µg/kg dw for acetone. All other VOCs were detected at concentrations less than 35 µg/kg dw. In subsurface sediment collected between RM 3.5 and RM 3.6 as part of the RI, high concentrations of cis-1,2-dichloroethene and vinyl chloride (200,000 and 60,000 µg/kg dw, respectively) were detected at depth (between 10 and 12 ft). These VOCs were not detected within the top 6 ft of sediment in this core.

Few VOCs were detected in seep water samples collected from Boeing Plant 2, Rhône-Poulenc, and at locations throughout the LDW as part of the RI; concentrations were low (≤ 40 µg/L). Numerous VOCs were frequently detected in seep water samples from GWI. VOCs with the highest concentrations were cis-1,2-dichloroethene (5,400 µg/L) and vinyl chloride (3,500 µg/L).

Porewater samples were collected and analyzed for VOCs at Boeing Plant 2/Jorgensen Forge and GWI as part of the RI. At GWI, cis-1,2-dichloroethene and vinyl chloride were detected in all 10 nearshore samples, at maximum concentrations of 2,900 and 2,500 µg/L, respectively. At Boeing Plant 2/Jorgensen Forge, VOCs were detected at low concentrations in porewater samples (≤ 13 µg/L).

4.3 COMPARISON OF DEPTHS OF PEAK PCB CONCENTRATIONS IN CORES WITH PREDICTIONS OF THE STM

This section combines information on depth patterns in sediment chemistry (from sediment cores) with net sedimentation and erosion estimates from the STM to determine whether vertical patterns of total PCB concentrations are consistent with the CSM and with the predictions of the STM. This analysis is focused on total PCBs because PCBs are persistent, have a relatively clear history of use, and are widespread. The FS will consider this type of analysis and other lines of evidence when evaluating remedial alternatives and remedial technologies. The FS will also include a site-specific analysis of areas with localized erosion (e.g., berthing activities and associated ship-induced bed scour).

The CSM for sediment transport was refined based on results of the STM, which generates estimates of net sedimentation and net erosion for each LDW model cell (Maps 3-4, 3-5, 3-6, and 3-9).[104] Results of the STM indicate that the LDW is net depositional over annual time scales, although rates of net sedimentation vary across the LDW from < 0.5 cm/year to approximately 140 cm/year (the latter value only in the Upper Turning Basin), with typical rates for most of the LDW between 0.5 and 3 cm/yr. The STM also identified areas of the LDW where erosion could occur during high-flow events.[105] According to the STM, high-flow events are unlikely to erode sediment to depths greater than approximately 22 cm, and net erosion up to this depth is expected in relatively small areas (Map 3-7). Other physical events that could potentially expose subsurface sediments deeper than approximately 22 cm include

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[104] The STM models conditions within 2,000 horizontal grid cells, with seven lateral (i.e., from bank to bank) grid cells at most locations in the LDW. Cell lengths (north-south axis) ranged from 20 to 150 m. For each model cell, the STM estimated annual average net sedimentation rates over a 30-year simulation and net erosion depths over the course of a high-flow event.

[105] High-flow events include above-average flows with return periods of 2 years (8,400 cfs), 10 years (10,800 cfs), and 100 years (12,000 cfs). The average LDW flow is 1,350 cfs. The results from the three high-flow event simulations were similar.

stationary tugboats using high thrust, dredge events, or construction events; these events are expected to be highly localized and of short duration. Results of an analysis of ship-induced bed scour conducted as part of the STM (see Section 3.1.7) indicated that erosive forces from tugboats transiting the navigation channel tend to repeatedly mix the upper 1 to 2 cm of sediment in the areas frequented by these ships. Ship-induced mixing was evaluated only for ships traveling within the navigation channel, not for ships maneuvering in berthing areas (QEA 2008).

As described in Section 3.1.4, net sedimentation rates were also estimated empirically using several lines of evidence. These empirically-derived sedimentation rates were compared with the sedimentation rates from the STM in Table 3-9. In general, net sedimentation rates estimated using the empirical approach agreed reasonably well with the model estimates, considering the different methods used to derive the estimates. One of the empirical lines of evidence used to estimate net sedimentation rates was the vertical profile of PCB concentrations in sediment cores. This line of evidence is discussed in greater detail below by comparing to STM-derived net sedimentation rates (in addition to the comparisons presented in Section 3.2.5).

Much of the sediment contamination in the LDW, and particularly PCB contamination, is believed to have originated from historical sources in the LDW (see Section 9). In undisturbed depositional areas with no ongoing or recent sources, PCB concentrations should be higher in deeper core intervals than in shallower intervals. In areas with little or no deposition, localized disturbances, or ongoing or recent secondary sources (e.g., erosion of contaminated soil), this pattern may be reversed, with higher PCB concentrations in the shallowest core intervals.

Assuming that an area is depositional and has not been disturbed, the depth[106] of the maximum total PCB concentration within a core should be a function of both the time since peak PCB use and release and the estimated rate of net sedimentation. In this analysis, the maximum total PCB concentration within a core is referred to as the peak total PCB concentration. To investigate this CSM, the expected depth of peak total PCB concentration was estimated for each core. This depth within a core was calculated as described in Equation 4-2:

$$D = (Tc - Tm) \times S$$ **Equation 4-2**

Where:

D = depth of peak total PCB concentration (cm)
Tc = year of core collection

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[106] Sampling intervals in cores in the RI dataset were based on recovered depths for all cores collected for the RI, and for most cores collected as part of other sampling events.

Tm = assumed year of maximum concentration in surface sediment, corresponding to the assumed peak in PCB use and releases to the LDW

S = net sedimentation rate (cm/yr) estimated from the STM for the cell containing the core[107]

General uncertainties associated with estimating the depth of the peak total PCB concentration using Equation 4-2 include uncertainties in the net sedimentation rate estimated by the STM and uncertainty in the estimate of the year of the peak release of PCBs. In addition, there is uncertainty associated with identifying the exact depth of the peak total PCB concentration within a core because of compositing within each core section. Uncertainty is particularly high at locations where the core intervals analyzed were 3 ft or greater and is lowest at locations where the core was sectioned into 0.5-ft intervals. Location-specific uncertainties include the possibility of sediment disturbance near berthing areas or local structures, and the potential for PCB releases to continue after the peak use/release date.

To address the uncertainty in the year of maximum historical PCB releases to the LDW, a range of estimated depths of the peak total PCB concentration was calculated for each core corresponding to a range of years during which maximum releases may have occurred (i.e., estimated depths within each core were calculated assuming maximum PCB releases in 1960, 1965, and 1974). Three years were assessed to acknowledge the uncertainty in the year of maximum PCB release. Peak PCB concentrations were found in Puget Sound sediment cores at depths corresponding to between 1960 and 1970 (Van Metre and Mahler 2004; Lefkovitz et al. 1997); the use of PCBs was banned in 1978 and they were phased out over the next few years.[108] The specific years selected were based on the following rationale that applies throughout the LDW site: 1960 is the earliest justifiable year based on the Puget Sound core data, 1965 is mid-way through the 1960s, and 1974 was the year of a PCB spill in Slip 1 and PCBs were still being produced with the potential for release into the environment.

These estimated depths were then compared with the actual depth of the interval with the peak total PCB concentration in each core. If the depth of the interval with the peak total PCB concentration in a core was within the range of estimated depths or deeper than the range of estimated depths, then the core was considered to be consistent with the CSM and with the predictions of the STM. If peak depths estimated by the STM for the three years assumed to be associated with maximum PCB releases were all deeper than the core depth with the peak total PCB concentration, then the core was considered inconsistent with the CSM and the predictions of the STM (i.e., actual net

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[107] If a core was collected in an area outside the boundaries of the STM, the net sedimentation rate of the nearest cell was used.

[108] Even after PCBs were phased out, releases could continue from secondary sources (e.g., contaminated soils, paints).

sedimentation rates must be lower than those estimated by the STM). See Maps 4-69a through 4-69d for examples of consistent and inconsistent cores.

Detailed results of this analysis for each individual core are presented in Appendix F. Of the 366 cores available in the RI dataset, 157 cores were used in the analysis and 209 cores were not used because the type of information needed for the analysis was not available for those cores (Table 4-94). Cores were excluded if at least one of the following conditions was met: 1) only one core interval was analyzed for total PCBs, 2) no core interval was analyzed within the depth range of the expected peak, 3) PCBs were not detected in any core interval, or 4) the sediment was disrupted by dredging prior to sampling.

Table 4-94. Results of the analysis comparing expected and actual depths of peak total PCB concentrations in subsurface sediment

DETERMINATION	REASON	NUMBER OF CORES	
		ALL CORES	EXCLUDING CORES WITH LARGER INTERVALS
Consistent with STM	Peak was as deep or deeper than expected depth range.	110 (70%)	86 (65%)
Inconsistent with STM	Peak was shallower than expected depth range.	47 (30%)	47 (35%)
Total number of cores included		157	133
Excluded from analysis	At least one of the following conditions was associated with a core: 1) only one core interval was analyzed for total PCBs, 2) no core interval was analyzed within the depth range of the expected peak, 3) PCBs were not detected, or 4) dredging occurred prior to sampling.	209	233[a]
Total number of cores		**366[b]**	**366[b]**

[a] Twenty-four cores were excluded from the analysis because they had ≥ 3 ft sampling intervals in the portion of the core near the estimated and actual peak depths. These cores are identified in Tables F-1 and F-2 of Appendix F.

[b] Four of the cores were collected outside of the STM boundaries so the net sedimentation rate of the nearest cell was used.

PCB – polychlorinated biphenyl

STM – sediment transport model

Of the 157 cores included in the analysis, 110 cores (70%) had peak total PCB concentrations at depths equal to or greater than the estimated depths, consistent with the predictions of the STM. Forty-seven cores (30%) had maximum total PCB concentrations that were shallower than the estimated depth range based on net sedimentation rates from the STM, assumed maximum PCB releases to the LDW in the 1960s and 1970s, and minimal localized disturbances. Thirty-two of these forty-seven cores were located in EAAs with either very low net sedimentation rates or in areas where source investigations have suggested the potential for ongoing releases of PCBs.

Uncertainty in determining whether cores were consistent or inconsistent was higher in cores with samples composited over larger intervals (i.e., ≥ 3 ft). To assess this uncertainty, 24 cores with samples composited over ≥ 3 ft intervals in the portion of the core where the estimated and actual peak depths occurred were removed to assess the influence on the results. Cores were retained in this uncertainty analysis if the intervals ≥ 3 ft were substantially deeper than the estimated and actual peak depths because these data were not expected to affect the outcome of the analysis. Note that these 24 cores were consistent with the STM; however, because of the larger intervals, their intrepretation is more uncertain. When the 24 cores were removed from the analysis, for a total of 133 cores in the analysis, 86 cores (65%) were considered consistent with the predictions of the STM, and 47 cores (35%) were considered inconsistent.

The results of the initial analysis using all 157 cores are discussed in more detail in the following sections for each of the three reaches defined in the CSM: Reach 1 – from RM 0.0 to RM 2.2; Reach 2 – from RM 2.2 to RM 4.0; and Reach 3 – from RM 4.0 to RM 4.8.

4.3.1 Reach 1 – RM 0.0 to RM 2.2

Based on the results of the STM, Reach 1 was characterized as net depositional on annual time scales in both the navigation channel and the adjacent bench areas. The sedimentation rates in the navigation channel and bench areas were generally classified as intermediate net depositional (1 to 2 cm/yr) within the first RM, and as intermediate to high depositional (1 to 3 cm/yr) in the remaining portion of this reach. Some relatively small areas that were classified as lower net depositional (0 to 1 cm/yr) are also present in this reach (Map 3-9).

In general, the potential for re-exposing buried sediments in this reach as a result of scour during high-flow events is considered minimal. The STM identified the potential for relatively low bed scour (< 2 cm) during a 100-year event in a small area between RM 0.8 and RM 0.9 and in several areas between RM 1.8 and RM 2.2 (Map 3-7). Ship-induced mixing of the surficial bed layer from ships traveling within the navigation channel was estimated to extend to average depths of about 1 to 2 cm in the bench areas and less than 1 cm in the navigation channel.

Maps 4-69a through 4-69d present an overlay of sediment core locations on a map depicting estimated average annual net sedimentation rates in the LDW. These maps show the depth interval of the actual peak total PCB concentration in subsurface sediment relative to the estimated depth of the peak concentration and indicate whether the actual depth is consistent with (the same depth or deeper than) or shallower than the estimated depth.

Most patterns of total PCB concentrations as a function of depth within the cores in this reach were consistent with the net sedimentation rates estimated by the STM and estimated period of peak PCB releases to the LDW. Three of the fifty-nine cores in this reach with data sufficient to assess consistency with the predictions of the STM were field replicates; thus, those fifty-nine cores represent fifty-six locations. Of the 56 locations with sufficient core data, 43 (77%) had peak total PCB concentrations as deep or deeper than expected, and 13 locations (23%) had peak total PCB concentrations shallower than expected (Table 4-95). Examples of cores in which peak total PCB concentrations were consistent with the STM can be found at locations throughout the reach, including cores along the eastern bench, along the western bench, and in the navigation channel (Maps 4-69a and 4-69b).

Table 4-95. Results of the analysis comparing expected and actual depths of peak total PCB concentrations in subsurface sediment by river reach

		NUMBER OF CORES		
RIVER REACH	AREA WITHIN REACH	CONSISTENT WITH STM	INCONSISTENT WITH STM	EXCLUDED FROM ANALYSIS[a]
RM 0 to RM 2.2	EAA 1 (Duwamish/Diagonal)	13	0	0
	rest of reach	31	15	49
	Subtotal	**44 (43)[b]**	**15 (13)[c]**	**49**
RM 2.2 to RM 4	EAA 2 (RM 2.2)	0	1	0
	EAA 3 (Slip 4)	6	0	0
	Boeing Plant 2/Jorgensen Forge EAA	47	19	66
	EAA 5 (Terminal 117)	3	8	9
	EAA 6 (RM 3.8 to 3.9)	0	1	0
	rest of reach	9	3	40
	Subtotal	**65**	**32**	**115**
RM 4 to RM 4.8	entire reach	1	0	45
	subtotal	**1**	**0**	**45**
Total		**110**	**47**	**209**

[a] Cores were excluded from this analysis if at least one of the following conditions was met: 1) only one core interval was analyzed for total PCBs, 2) no core interval was analyzed within the depth range of the expected peak, 3) PCBs were not detected, or 4) dredging occurred prior to sampling.

[b] One core of the 44 cores was a field replicate.

[c] Two cores of the 15 cores were field replicates.

EAA – early action area
PCB – polychlorinated biphenyl
RM – river mile
STM – sediment transport model

There were 13 locations (23%) at which the peak total PCB concentration was in a shallower core interval than expected. Six of these locations were on the eastern bench, six were on the western bench, and one was in the channel. Differences between the expected and actual depths of the peak total PCB concentration may be a result of: 1) uncertainty in the sedimentation rates or scour depth predicted by the STM, particularly if the location was just outside of the modeling area or in areas with in-water structures or bathymetric features not captured by the model, 2) the presence of localized, ongoing sources of PCBs, or 3) the occurrence of localized disturbance events sufficient to expose subsurface sediment.

Ten of the thirteen locations that did not fit the expected pattern had relatively high PCB concentrations (> 210 µg/kg dw [the 75th percentile]) in the uppermost core interval. These 10 locations were situated throughout the reach, mostly close to the shoreline. Most of the 10 locations were near an outfall of some kind; in addition, one was outside of the modeling area and four were near berthing areas. Potential sources of PCBs have not been characterized in detail in these areas.

Total PCB concentrations in cores from the remaining 3 of the 13 locations were ≤ 75 µg/kg dw throughout the core profile. Two of these locations were on the east side of the navigation channel (near RM 1.1 and at the mouth of Slip 3) and one location was on the west side of the channel (near RM 1.4). Therefore, although the expected and actual peak total PCB concentrations were not at the same depth, differences in concentrations among the intervals within cores were relatively small.

In summary, data from a majority of the cores (77%) in Reach 1 were consistent with the predictions of the STM, with the peak PCB concentration as deep or deeper than the estimated depth. In the remaining 23% of the cores in which the peak PCB concentration was in a shallower interval than estimated, it is uncertain if the discrepancy was a result of uncertainties in the STM, the occurrence of localized disturbances, low PCB concentrations throughout the core resulting in relatively small differences in among the intervals, or the presence of ongoing sources. In such areas, it will be important to evaluate additional lines of evidence to predict changes in chemical concentrations in the future.

4.3.2 Reach 2 – RM 2.2 to RM 4.0

Reach 2 was characterized as net depositional, with high spatial variability in net sedimentation rates, ranging from 0 to approximately 11 cm/year (Maps 4-69c and 4-69d). Some of the benches within the reach were characterized as having high net sedimentation rates (> 3 cm/yr), including both eastern and western benches between RM 2.0 and RM 2.6 and the western bench between RM 3.3 and RM 3.6. This reach also includes several areas with the lowest net annual sedimentation rates (0 to 0.5 cm/yr). Estimated scour depths were also variable, with a maximum net erosion depth of 21 cm in a small area near RM 3.6. Scour depths were generally higher in the navigation channel than in the bench areas. Ship-induced mixing of the surficial bed layer from ships traveling within the navigation channel is expected to be minimal, potentially extending to average depths of less than 1 cm in the bench areas and less than 0.1 cm in the navigation channel.

As in Reach 1, most patterns of total PCB chemical concentrations as a function of depth within the cores in Reach 2 were consistent with the net sedimentation rates estimated by the STM and the assumed peak usage of PCBs (Maps 4-69c and 4-69d).

Of the 97 locations in this reach with data sufficient to assess consistency with the predictions of the STM, 65 (67%) had peak total PCB concentrations as deep or deeper than expected based on the estimated net sedimentation rate over the time period since the expected peak usage and release of PCBs (Table 4-95). These cores were distributed throughout the reach (Maps 4-69c and 4-69d).

There were 32 locations (33%) that had peak total PCB concentrations in a shallower interval than expected. Eight of the cores were collected near T-117 and nineteen of the cores were collected in the Boeing Plant 2/Jorgensen Forge EAA near RM 3.6. The remaining five cores were located near RM 2.2 (in EAA 2) and near RM 2.8, RM 3.3, RM 3.8, and RM 3.9. Most of these cores were also located near outfalls. Potential sources have been characterized in the EAAs, but not in all of the other areas.

Twenty-seven of the thirty-two cores in Reach 2 that did not fit the expected pattern had relatively high total PCB concentrations (> 210 µg/kg dw) in the uppermost core interval analyzed. The five cores that did not have total PCB concentrations greater than 210 µg/kg dw were located near the inlet at RM 2.2, in the channel near RM 3.3, in the T-117 EAA near RM 3.7, and in the Boeing Plant 2/Jorgensen Forge EAA (two cores).

In summary, data from 65 (67%) of the cores in this reach were consistent with the predictions of the STM, with the peak PCB concentration as deep or deeper than the estimated depth. However, at 32 (33%) of the core locations, peak PCB concentrations were detected in a shallower core interval than that estimated by the STM. At these 32 locations, it appears that either net sedimentation is lower than expected, localized disturbances are occurring, or ongoing sources may be present. All but three of these 32 cores were located in EAAs, and all but one of the three cores outside EAAs were near outfalls.

4.3.3 Reach 3 – RM 4.0 to RM 4.8

This reach was characterized as net depositional, with relatively high net sedimentation rates: 2 to 3 cm/year in Slip 6; 4 to 23 cm/year between RM 4.0 and RM 4.5, and up to 151.5 cm/year in the Upper Turning Basin, which is designed to act as a trap for much of the coarser fraction of the bed load entering the LDW from upstream. Episodic erosion may occur during high-flow events in some areas of the reach (up to approximately 21 cm in one small area at RM 4.6), but most of the reach is not expected to experience scour greater than 10 cm in depth during high-flow events. Ship-induced mixing of the surficial bed layer from ships traveling within the navigation channel is expected to be minimal.

Of the 47 cores in this reach, only one location (in Slip 6) had sufficient data to assess consistency with the predictions of the STM using this method (Table 4-95). At this location in Slip 6, peak total PCB concentrations were at a depth in the core consistent with expectations based on the STM and peak PCB usage.

4.3.4 Summary of comparison of depths of peak PCB concentrations in cores with predictions of the STM

A total of 157 cores from throughout the LDW were evaluated for consistency with the predictions of the STM. Most (70%) of the cores were consistent with the assumptions of the analysis, with peak PCB concentrations as deep or deeper than those estimated by the STM. The remaining 30% of the cores had peak concentrations that were shallower than expected assuming net sedimentation rates from the STM, peak PCB usage and release in the 1960s and 1970s, and minimal localized disturbances. Therefore, one or more of these assumptions were likely not valid at these particular locations. The principal limitations of this analysis include uncertainties in the net sedimentation rate estimated by the STM, the estimate of the peak year of PCB release in the LDW, and the identification of the depth of peak PCB concentrations in sediment cores with larger intervals. An uncertainty analysis was conducted to assess the influence of the larger intervals. When the 24 cores were removed from the analysis (because they had samples composited over ≥ 3 ft in the interval with the actual or predicted peak), 86 cores (65%) were considered consistent with the predictions of the STM, and 47 cores (35%) were considered inconsistent.

The FS will evaluate other lines of evidence and will include a more detailed analysis of core locationsthat were inconsistent with the predictions of the STM to look for the presence of site features or used that may account for these inconsistencies. Eighteen percent of the cores with shallower than expected peaks were located in EAAs.

4.4 SUMMARY

This section provides a summary of the nature and extent of contamination in the LDW based on a comprehensive chemical dataset for surface sediment, subsurface sediment, tissue, surface water, seep water, and porewater.

Surface sediment samples from 1,484 locations within the LDW were used to evaluate the nature and extent of contamination. Chemical contamination of surface sediment in the LDW is characterized by a discrete set of localized areas with high chemical concentrations (hot spots) separated by relatively large areas with much lower contamination.

Table 4-96 presents summary statistics for selected risk driver chemicals identified in the risk assessments and also lists locations with the highest concentrations. The SWAC provides an indication of average concentrations within the LDW that is not biased by sampling density, whereas the area-based 95th percentile identifies areas with the highest concentrations in the waterway (95% of the LDW, by area, has lower concentrations). As shown on Maps 4-70a through 4-70c, areas with the highest chemical concentrations of risk driver chemicals tend to be co-located. Many of the locations with the highest concentrations are within areas that have been identified as EAAs (see Appendix I).

Table 4-96. Summary of LDW surface sediment data

CHEMICAL	UNIT	95TH PERCENTILE[a]	50TH PERCENTILE[a]	SWAC	AREAS WITH HIGHEST CONCENTRATIONS[b]
Total PCBs	µg/kg dw	810	110	350	Duwamish/Diagonal EAA RM 1.0 in navigation channel EAA 2 Slip 4 EAA Boeing Plant 2/Jorgensen Forge EAA T-117 EAA Norfolk EAA
Arsenic	mg/kg dw	25	11	15	RM 3.8 EAA near RM 0.1 on the east side Duwamish/Diagonal EAA Slip 1 RM 1.3 to RM 1.5 on the west side Slip 3 Boeing Plant 2/Jorgensen Forge EAA Between RM 3.7 and RM 3.9 EAA on the east side
cPAHs	µg/kg dw	1,100	260	380	T-117 EAA Slip 4 EAA RM 3.8 EAA near RM 0.6 on the west side RM 0.0 near Harbor Island Duwamish/Diagonal EAA
BEHP	µg/kg dw	990	200	380	Duwamish/Diagonal EAA EAA 2 Slip 4 EAA Boeing Plant 2/Jorgensen Forge EAA Norfolk EAA
Dioxin and furan TEQ	ng/kg dw	490	10	nc	RM 1.4 to RM 1.5 on the west side EAA 2 Duwamish/Diagonal EAA near RM 1.0 in the navigation channel Boeing Plant 2/Jorgensen Forge EAA
Cadmium	mg/kg dw	1.8	0.39	nc	Boeing Plant 2/Jorgensen Forge EAA Duwamish/Diagonal EAA Slip 1 Slip 4 RM 1.3 to RM 1.5 on the west side near RM 0.1 on the east side near RM 1.0 in the navigation channel EAA 2
Chromium	mg/kg dw	70	29		
Copper	mg/kg dw	147	52		
Lead	mg/kg dw	210	36		
Mercury	mg/kg dw	0.41	0.14		
Nickel	mg/kg dw	43	22		
Silver	mg/kg dw	1.7	0.3		
Vanadium	mg/kg dw	81	57		
Zinc	mg/kg dw	407	113		
TBT	µg/kg dw	250	22	nc	near RM 1.3 on the west side between RM 0.0 and RM 0.4
HPAH	µg/kg dw	13,300	1,985	nc	T-117 EAA Between RM 0.5 and RM 0.6 in the navigation channel and on the west side RM 3.8 EAA RM 2.7 on the west side near RM 1.0 on the east side Boeing Plant 2/Jorgensen Forge EAA Duwamish/Diagonal EAA Slip 1 Slip 4

CHEMICAL	UNIT	95TH PERCENTILE[a]	50TH PERCENTILE[a]	SWAC	AREAS WITH HIGHEST CONCENTRATIONS[b]
LPAH	µg/kg dw	2,300	230	nc	RM 2.7 on the west side T-117 EAA near RM 1.0 on the east side Boeing Plant 2/Jorgensen Forge EAA Duwamish/Diagonal EAA

[a] The 95th and 50th percentiles were calculated on an area basis for total PCBs, arsenic, cPAHs, and BEHP. For the remaining chemicals, the percentiles were calculated arithmetically using all values from the baseline surface sediment dataset. UCLs for total PCBs, arsenic, cPAHs, and dioxins and furans in surface sediment were calculated using the RI baseline dataset in Appendix C (Table C.3-8). These UCLs (2,500 µg/kg dw for total PCBs, 21 mg/kg dw for arsenic, 570 µg/kg dw for cPAHs, and 610 ng/kg dw for dioxin and furan TEQ) are presented as EPCs for the netfishing scenario in the HHRA (Appendix B), which uses the entire surface sediment dataset.

[b] Areas with the highest concentrations are listed but may not include all areas with concentrations greater than the 95th percentile; see individual maps for all locations.

BEHP – bis(2-ethylhexyl) phthalate
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CSO – combined sewer overflow
dw – dry weight
EAA – early action area
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LDW – Lower Duwamish Waterway
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
nc – not calculated
PCB – polychlorinated biphenyl
RM – river mile
SWAC – spatially weighted average concentration
T-117 – Terminal 117
TBT – tributyltin

Surface sediment data were compared with SMS criteria, and Thiessen polygons were used to estimate the areal extent of potential effects based on combined toxicity test results and surface sediment chemistry data. Using this approach, approximately 7% (34 ac) of the LDW area, had chemical concentrations or biological effects that exceeded CSL criteria and 18% (82 ac) of the LDW area had chemical concentrations or biological effects between the SQS and CSL criteria.

Subsurface sediment contamination was also characterized by many studies within the LDW. Many of the areas with high chemical concentrations in surface sediment also had high concentrations in subsurface sediment, although some areas with less contaminated surface sediment had contamination in the subsurface, particularly for PAHs.

A detailed analysis of the locations of peak concentrations of total PCBs, arsenic, and cPAHs within individual subsurface cores was conducted. At many locations, subsurface sediment at deeper intervals contained higher total PCB and cPAH concentrations than did sediment from the uppermost intervals, with some exceptions.[109]

The two highest dioxin and furan TEQs (194 and 136 ng/kg dw) in subsurface sediment were detected in samples collected deeper than 4 ft at locations LDW-SC20 (RM 1.0 in the navigation channel) and LDW-SC26 (RM 1.4 W), respectively. The next

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[109] The exceptions for total PCBs were near RM 1.0 in the navigation channel, EAA 2, the Slip 4 EAA, the T-117 EAA, and the Boeing Plant 2/Jorgensen Forge EAA. The exceptions for cPAHs were Slip 6, the Norfolk EAA, the Duwamish/Diagonal EAA, and near RM 0.5 on the west side. In all but one of the five areas with high arsenic concentrations in subsurface sediment, peak concentrations were at depth within the cores.

highest concentration in subsurface sediment (54.1 ng/kg dw) was detected in the uppermost core interval at LDW-SC29 (the embayment at RM 1.5 W) that also had a very high concentration (2,100 ng/kg dw) in surface sediment. Concentrations decreased with depth at this location.

Chemical patterns in sediment cores collected throughout the site were evaluated for consistency with the CSM and with the predictions of the STM. Based on the CSM, total PCB concentrations should be higher in deeper intervals of the subsurface sediment cores than in shallower intervals, except in areas with low net sedimentation, localized disturbances, or ongoing or recent secondary sources. Available core data collected throughout the LDW are generally consistent with this expected pattern: most (70%)[110] of the cores showed that peak total PCB concentrations are as deep or deeper than those estimated by the STM. The remaining 30% of the cores had peak concentrations that were shallower than expected. Most of these locations were either in areas just outside the STM modeling areas, in dredged areas, near berthing areas, or near locations with the potential for ongoing sources. This analysis focused on only one of the lines of evidence to be used in the evaluation of remedial alternatives in the FS. The FS will consider additional lines of evidence and will include a site-specific analysis of areas with localized erosion (e.g., berthing activities and associated ship-induced bed scour).

A large number of tissue composite samples have been collected in the LDW for various fish, crab, and invertebrate species. The locations and number of the four sampling areas for fish and crabs were selected primarily to reflect the longitudinal distribution of PCB concentrations in the sediment characterized by a rolling average PCB sediment concentration. In whole-body composite samples of fish, mean sampling area total PCB concentrations ranged from 71 µg/kg ww in juvenile chinook salmon in Area T1 to 4,300 µg/kg ww in shiner surfperch collected in 2004 in Area T2. The highest mean total PCB concentration was in crab hepatopancreas tissue (5,500 µg/kg ww in T4 in 2004); mean concentrations in edible crab meat were much lower, ranging from non-detect to 270 µg/kg ww. In general, total PCB concentrations in invertebrates (except crab hepatopancreas samples) tended to be lower than in fish, and concentrations in juvenile chinook salmon were lower than concentrations in English sole and shiner surfperch. Mean lipid-normalized total PCB concentrations in fish and crab tissue ranged from non-detected to 100 mg/kg-lipid, with no clear differences in concentrations among fish or crab species or tissue types (i.e., whole body, fillet, hepatopancreas, or edible meat). Mean lipid-normalized total PCB

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[110] An uncertainty analysis was conducted to assess the influence of the larger intervals. When the 24 cores were removed from the analysis (because they had samples composited over ≥ 3 ft in the interval with the actual or predicted peak), 86 cores (65%) were considered consistent with the predictions of the STM, and 47 cores (35%) were considered inconsistent.

concentrations in benthic invertebrates, clams, and mussels were generally lower than in fish and crab tissue, ranging from 3.3 to 41 mg/kg-lipid.

In general, the PCB congener patterns for each tissue type were similar, except for two clam samples from Slip 4 with higher contributions of less chlorinated PCB congeners. Of the three fish species analyzed for PCB congeners, shiner surfperch had the greatest variability in congener patterns among the individual samples. The mean pattern for whole-body fish samples in 2007 showed a small increase in the contribution from more highly chlorinated PCB congeners relative to the whole-body fish samples from 2004. The TEQ potencies were similar in the English sole and shiner surfperch tissues collected in 2004, 2005, and 2007. The Aroclor composition in samples collected from 2005 to 2007 appear to have had a higher proportion of the more highly chlorinated PCB congeners than those collected in 2004.

The Aroclor tissue data as a whole suggest total PCB concentrations in 2004 were higher than those in more recent years; total PCB concentrations appear to be decreasing based on Aroclor data. Trends based on the limited PCB congener data are more uncertain. The PCB congener data may suggest total PCB concentrations in English sole collected from Area T1 and possibly from Area T2 are decreasing, although the sample size is small. Total PCB concentrations (congener sums) in crabs collected from Areas T1 and T3 were also consistently lower in 2007 than 2004. The PCB congener data suggest temporal trends in total PCB concentrations in shiner surfperch are uncertain.

Differences in total PCB concentrations in English sole and shiner surfperch tissue among the four subareas were also evaluated. Area means were not significantly different in 2005 and 2007 for either wet-weight or lipid-normalized total PCB concentrations in English sole tissue. In 2004, significant differences were identified only between the area with the lowest mean (Area T4) and the areas with the highest or second highest mean (Area T1 or T2). For shiner surfperch, differences among areas were not significant in any year based on lipid-normalized total PCB concentrations. Wet-weight total PCB concentrations did not differ among areas in 2005 but were lower in Areas T1 and T4 than in Areas T2 and T3 in 2004 and 2007. Sufficient data for statistical analyses for Pacific staghorn sculpin were available for only 1 year (2004). Wet-weight and lipid-normalized total PCB concentrations in Pacific staghorn sculpin were significantly higher in Area T3 than in the other areas in 2004.

Mean total arsenic concentrations were highest in soft-shell clams (6.803 mg/kg dw in depurated samples). Mean inorganic arsenic concentrations, which are more relevant from a risk perspective, were also highest in soft-shell clams (3.37 mg/kg ww in depurated samples). Clam tissue had a very high percentage of inorganic arsenic (10 to 99%), much higher than fractions detected in tissue from other studies in Puget Sound. Regression analyses showed a statistically significant positive correlation between arsenic concentrations in benthic invertebrate tissue and co-located sediment; this relationship was used to calculate area-wide tissue concentrations for use in the ERA (Appendix A).

The highest cPAH concentrations in tissue were detected in clams, mussels, and benthic invertebrates at mean concentrations ranging from 15 to 44 µg/kg ww; cPAH concentrations in fish tissue were relatively low. Mean cPAH concentrations in English sole (whole-body tissue) were significantly lower in Area T4 than in Areas T1 and T2. For shiner surfperch, mean concentrations were significantly lower in Area T4 than in Area T3. The cPAH SWAC in surface sediment was also significantly lower in Area T4 than in any other areas.

The highest mean concentrations of metals (including TBT) were detected in invertebrates (i.e., clams, crabs, benthic invertebrates, and amphipods). Other than PAHs, which were detected in most tissue samples, SVOCs were infrequently detected in tissue. DDT and chlordane compounds were detected in tissue samples, but there is high uncertainty in these data because of analytical interference from PCBs.

To determine concentrations of various chemicals in surface water in the LDW, grab samples were collected and SPMDs were deployed in the mid 1990s as part of the WQA by King County**.** In 2005, surface water grab samples were also analyzed for PCB congeners. Total PCB congener concentrations in grab samples collected from two locations in the LDW on four sampling dates in 2005 ranged from 0.000132 to 0.003211 µg/L. Concentrations were lowest at all locations in November 2005, when water flow was the highest.

In surface water grab samples, mean dissolved and total arsenic concentrations were higher at RM 1.1 and RM 1.9 than at RM 4.9. Mean concentrations of chromium, copper, lead, nickel, vanadium, and zinc in unfiltered samples and copper, lead, and zinc in filtered samples were higher in samples from RM 4.9 than in samples from RM 1.1 and RM 1.9. SVOCs were detected infrequently in surface water samples collected from the LDW. Calculated PAH and organochlorine pesticide concentrations in surface water based on the SPMD data ranged from 0.0000029 to 0.030 µg/L for PAHs and from 0.000000034 to 0.000033 µg/L for pesticides.

Seep water chemical data have been collected at four sites along the LDW, including Rhône-Poulenc, Boeing Plant 2, T-117, and GWI as well as during an LDW-wide seep investigation conducted for the RI. In seep water samples, PCBs were detected in a filtered sample collected from one of the RI locations (in the inlet at RM 2.2). The highest dissolved arsenic concentrations in seep water (72.4 and 253 µg/L) were detected in the embayment near RM 1.5 on the west side and in Slip 1, respectively. These areas also had some of the highest arsenic concentrations in surface sediment. An unfiltered seep water sample collected near Slip 6 had the highest concentrations of cadmium, chromium, copper, nickel, and vanadium. SVOCs were detected infrequently in seep water samples collected from the LDW. One organochlorine pesticide (heptachlor epoxide) was detected at one seep location in both filtered and unfiltered samples.

Few VOCs were detected in seep water samples collected from Boeing Plant 2, from Rhône-Poulenc, and at locations throughout the LDW; concentrations were ≤ 40 µg/L. VOCs were frequently detected in seep water samples collected during various GWI investigations. VOCs with the highest concentrations were cis-1,2-dichloroethene (5,400 µg/L) and vinyl chloride (3,500 µg/L).

Porewater samples collected from non-dredged sediment have been analyzed in four separate sampling events (EPA SI, PSDDA 1999, Rhône-Poulenc, and the RI). As part of the EPA SI, 15 unfiltered porewater samples were collected from throughout the LDW and analyzed for metals; seven metals, including arsenic, were detected in these samples. TBT was also analyzed in porewater as part of the EPA SI and the PSDDA 1999 sampling event. Only one of the 35 porewater samples had a TBT concentration (0.55 µg/L) that was greater than the DMMP SL (0.15 µg/L). Most SVOCs were detected infrequently, except BEHP in the Rhône-Poulenc study, which was detected in 56% of porewater samples analyzed. Organochlorine pesticides and PCBs were not analyzed in porewater.

Porewater samples were collected and analyzed for VOCs at the Boeing Plant 2/ Jorgensen Forge EAA and at GWI. At GWI, cis-1,2-dichloroethene and vinyl chloride were detected in all 10 nearshore samples, at maximum concentrations of 2,900 and 2,500 µg/L, respectively. At the Boeing Plant 2/Jorgensen Forge EAA, VOCs were detected in porewater samples at concentrations less than 13 µg/L.

5 Summary of the Baseline Ecological Risk Assessment

The baseline ERA estimated risks for the benthic invertebrate, fish, and wildlife species that may be exposed to chemicals in sediment, water, and aquatic biota in the LDW. This assessment was based on historical data (used in the Phase 1 ERA) (2003a) and sediment and tissue chemistry data collected as part of the RI. This section summarizes the problem formulation, exposure assessment, effects assessment, risk characterization, uncertainty analysis, and selection of risk drivers for ecological receptors. The complete ERA is included as Appendix A to this RI.

5.1 PROBLEM FORMULATION

The problem formulation of the ERA established the overall scope of the assessment. The problem formulation included a description of the data available for conducting the ERA, the suitability of the data for risk assessment purposes, and a risk-based screening process that allowed the risk assessment to focus on chemicals of potential concern (COPCs) and eliminate chemicals that posed minimal risks to ecological receptors.

Data used in the ERA consisted largely of:

- Surface sediment (uppermost 0 to 15 cm) chemistry data[111]
- Tissue chemistry data for benthic invertebrates (including benthic infauna and epifauna, crabs, clams, and mussels), English sole, Pacific staghorn sculpin, shiner surfperch, and juvenile chinook salmon
- Site-specific sediment toxicity test results
- Site-specific evaluation of imposex in field-collected gastropods
- Sediment porewater chemistry data

Because it is generally considered impractical to evaluate risks for every potentially exposed species, it is standard ERA practice and consistent with the SMS to focus on representative receptor species that typify groups of organisms with specific exposure pathways. One objective of selecting representative receptors is to choose species for which the risk conclusions will be protective or representative of other species that were not explicitly evaluated. For example, an assessment of risks to great blue herons would be assumed to be protective of all wading birds that eat fish because of the higher exposure potential of great blue herons. Receptors may also be selected for risk

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[111] The ERA was conducted using the baseline surface sediment dataset, which included sediment data collected prior to early actions in the LDW. Since these data were collected, early actions in the LDW have been conducted at two locations (Duwamish/Diagonal and the BDC south SD in the Norfolk area). Therefore, the risks discussed in the ERA may not be representative of current risks at areas where cleanup has already occurred.

analysis if the species is highly valued by society (e.g., endangered or threatened species) or if the species is particularly sensitive to COPCs. Representative receptors of concern (ROCs) selected for the baseline ERA were the benthic invertebrate community, crabs, English sole, Pacific staghorn sculpin, great blue heron, spotted sandpiper, osprey, river otter, and harbor seal (Table 5-1).

Table 5-1. ROCs selected for the LDW and a summary of the considerations for selection

ROC	Exposure Route	Ecological Significance	Societal Significance	Site Use	Exposure Data Availability	Sensitivity
Benthic invertebrate community	direct contact, diet, sediment ingestion	food source for other invertebrates, fish, and mammals; nutrient cycling	target community for protection in the development of numerical sediment quality criteria	present year-round; multiple life stages	abundant surface sediment data available	because of the diversity of organisms in this ROC group, the range of sensitivities is represented
Crabs	direct contact, diet, incidental sediment ingestion	higher-trophic-level benthic invertebrate	some recreational and commercial value	present year-round; multiple life stages	site-specific tissue data available	effects data are available for decapods; unknown relative sensitivity of crabs
Pacific staghorn sculpin	incidental sediment ingestion; preys on both invertebrates and other fish; thus potential for elevated exposure via bioaccumulation because of trophic position	serves as prey for higher-trophic-level species such as birds and mammals	considered to be a nuisance by anglers	adults and juveniles present year-round; may spawn in the LDW	site-specific tissue data and prey tissue data available	Effects data are available for other fish species; unknown relative sensitivity of sculpin
English sole	direct contact, diet, incidental sediment ingestion	important prey item for birds and fish; key benthic predator	some recreational and commercial value	juveniles present year-round; adults present except during spawning migrations to Puget Sound	site-specific tissue and prey tissue data available	NMFS data suggest that they are as sensitive as other flatfish species (Myers et al. 1998b)
Juvenile chinook salmon	diet	important prey item for birds and fish; seasonally one of the most abundant juvenile salmonids in the LDW	threatened and endangered species;[a] returning adults important to commercial, sport, and tribal fisheries	generally present April to July; most estuary-dependent juvenile salmonid	site-specific tissue and prey tissue data available	believed to be sensitive to a wide range of COPCs
Great blue heron	preys on fish; thus potential for elevated exposure via bioaccumulation because of trophic position	high on food chain	valued by society	present year-round; feed in LDW	site-specific data available for some food resources	effects data available for other bird species; unknown relative sensitivity of heron
Osprey	preys on fish; thus potential for elevated exposure via bioaccumulation because of trophic position	top of food chain	valued by society; protected under migratory bird treaty	nests along the LDW and forages at shallow depths in the LDW	site-specific data available for food resources	effects data available for other bird species; unknown relative sensitivity of osprey
Spotted sandpiper	diet, incidental sediment ingestion	important role as an intermediate predator	protected under migratory bird treaty	present June to September; nests along the LDW	site-specific data available for food resources	effects data available for other bird species; unknown relative sensitivity of sandpiper.

ROC	Exposure Route	Ecological Significance	Societal Significance	Site Use	Exposure Data Availability	Sensitivity
River otter	preys on fish; thus, potential for elevated exposure via bioaccumulation because of trophic position	top of food chain	valued by society	present year-round	site-specific data available for food resources	some mustelids shown to be highly sensitive to some chemicals (e.g., PCBs)
Harbor seal	preys on fish; thus potential for elevated exposure via bioaccumulation because of trophic position	top of food chain	protected under Marine Mammal Act	infrequent	site-specific data available for food resources	pinnipeds suspected to be sensitive to some chemicals (e.g., PCBs)

[a] Species listed under the Endangered Species Act.

COPC – chemical of potential concern
LDW – Lower Duwamish Waterway
NMFS – National Marine Fisheries Service
PCB – polychlorinated biphenyl
ROC – receptor of concern

In addition, juvenile chinook salmon was selected as an ROC because this species is federally protected and uses the LDW during outmigration to Puget Sound.

For each ROC selected, COPCs were identified through a conservative, risk-based screening process using maximum exposure concentrations and the SQS for benthic community or no-observed-adverse-effect levels (NOAELs) from the scientific literature for other receptors. COPCs identified included: 46 chemicals for the benthic invertebrate community (including TBT, metals, and PCBs and other organic compounds) (Table 5-2); 2 chemicals for crabs (total PCBs and zinc); 6 chemicals for at least one fish ROC (arsenic, cadmium, copper, total PCBs, TBT, and vanadium) (Table 5-3), and 12 chemicals for at least one wildlife ROC (arsenic, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, total PCBs, zinc, and vanadium) (Table 5-4). COPCs were evaluated only in the uncertainty analysis if there was uncertainty regarding their presence at concentrations of concern (i.e., chemicals were never detected in tissue, but RLs were above the screening criteria) or if they did not have effect-level toxicity information. Risks from organochlorine pesticides were also assessed in the uncertainty analysis. Detected concentrations of organochlorine pesticides in some sediment samples and in all tissue samples collected in 2004 were qualified as estimates (JN-qualified) because of probable analytical interference from PCBs, resulting in a high bias in concentration and a tentative identification (Windward 2005a, b).

Table 5-2. COPCs evaluated in the risk characterization for the benthic invertebrate community and for crabs

COPC	Benthic Invertebrate Community	Crabs
Metals		
Arsenic	X	
Cadmium	X	
Chromium	X	
Copper	X	
Lead	X	
Mercury	X	
Nickel	X	
Silver	X	
Zinc	X	X
Organometals		
TBT	X	
PAHs		
Acenaphthene	X	
Anthracene	X	
Benz(a)anthracene	X	
Benzo(a)pyrene	X	
Benzo(g,h,i)perylene	X	

COPC	BENTHIC INVERTEBRATE COMMUNITY	CRABS
Chrysene	X	
Dibenzo (a,h)anthracene	X	
Fluoranthene	X	
Fluorene	X	
Indeno (1,2,3,-c,d)pyrene	X	
Naphthalene	X	
Phenanthrene	X	
Pyrene	X	
Total benzofluoranthenes	X	
HPAH	X	
LPAH	X	
Phthalates		
BBP	X	
BEHP	X	
Dimethyl phthalate	X	
Other SVOCs		
1,2,4-Trichlorobenzene	X	
1,2-Dichlorobenzene	X	
1,4-Dichlorobenzene	X	
2-Methylnaphthalene	X	
4-Methylphenol	X	
2,4-Dimethylphenol	X	
Benzoic acid	X	
Benzyl alcohol	X	
Dibenzofuran	X	
Hexachlorobenzene	X	
n-Nitrosodiphenylamine	X	
Pentachlorophenol	X	
Phenol	X	
VOCs		
cis-1,2-Dichloroethene	X	
PCBs		
Total PCBs	X	X
Organochlorine Pesticides		
Total DDTs	X	
Total chlordane	X	

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin
VOC – volatile organic compound

Table 5-3. COPCs selected for fish ROCs

COPC	JUVENILE CHINOOK SALMON	ENGLISH SOLE	PACIFIC STAGHORN SCULPIN
Arsenic	X	X	X
Cadmium	X	X	X
Copper	X	X	X
Vanadium	X	X	X
TBT			X
Total PCBs		X	X

COPC – chemical of potential concern
PCB – polychlorinated biphenyl
ROC – receptor of concern
TBT – tributyltin

Table 5-4. COPCs selected for wildlife ROCs

COPC	SPOTTED SANDPIPER	GREAT BLUE HERON AND OSPREY	RIVER OTTER	HARBOR SEAL
Arsenic	X		X	
Cadmium	X			
Chromium	X	X		
Cobalt	X		X	
Copper	X			
Lead	X	X		
Mercury	X	X	X	X
Nickel	X			
Selenium	X		X	
Vanadium	X			
Zinc	X			
Total PCBs	X	X	X	X

COPC – chemical of potential concern
PCB – polychlorinated biphenyl
ROC – receptor of concern

No quantitative ecological risks were estimated for dioxins and furans because tissue samples from the LDW were not available for dioxins and furans at the time the risk assessments were completed.[112] Therefore, the level of ecological risk from dioxins and

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[112] In the human health risk assessment, it was assumed that risks associated with exposure to dioxins and furans are unacceptable, although tissue data were not available for specific exposure estimates. Dioxins and furans were assumed to be a risk driver based on human health risks associated with both the seafood consumption and direct contact pathways.

furans is unknown. Although risk from exposure to dioxin-like PCB congeners was evaluated for English sole and wildlife receptors, risks associated with exposure to all dioxin-like chemicals were not quantified. Thus, the combined exposure from all chemicals that bind with the Ah receptor was likely underestimated, although it is uncertain whether the inclusion of dioxins and furan exposure would have changed risk conclusions. The decision to not analyze dioxins and furans in tissue was made based on the difficulties associated with assessing site-specific risks from dioxins and furans, the need for a large background dataset with which to compare site-specific data, and the paucity of background tissue data in the Puget Sound area. Risk management decisions to address dioxin and furan contamination in LDW sediment will be based on relevant guidance and regulations, including those specifically related to the role of background-level contamination in the CERCLA process (EPA 2002a). Remedial decisions to address dioxin and furan contamination in sediment will be made by EPA and Ecology as part of the FS process.

The problem formulation also presented the conceptual site models for the ROCs (Figures 5-1 and 5-2). Conceptual site models identify and describe pathways in which ROCs may be exposed to COPCs associated with sediment within the LDW. The pathways evaluated in the ERA included both direct exposure through sediment and water and indirect exposure through the ingestion of prey from the LDW. The potential exposure pathways of sediment-associated chemicals to upper-trophic-level ROCs in the LDW were presented in a generalized food web diagram (Figure 5-3).

Assessment and measurement endpoints were also identified in the problem formulation (Table 5-5). Survival, growth, and reproduction were included as endpoints for all ROCs except juvenile chinook salmon. Assessment endpoints for juvenile chinook salmon included survival and growth; studies reporting reproductive endpoints were not used to derive reproductive TRVs for juvenile chinook salmon because of their life stage at the time of exposure (migrating juveniles) and because their exposure to LDW sediments as adults is limited.[113] Biomarker, behavioral, and histological endpoints were not included as assessment endpoints. Typically, ERAs focus on ecological effects at the individual level or higher (i.e., population level). In this way, the emphasis is placed on endpoints that integrate an overall response by an organism, rather than indicators of a biochemical response that may or may not result in an ecologically relevant effect. The assessment scale for each ROC was identified based on the potential exposure area for the ROC (Table 5-5).

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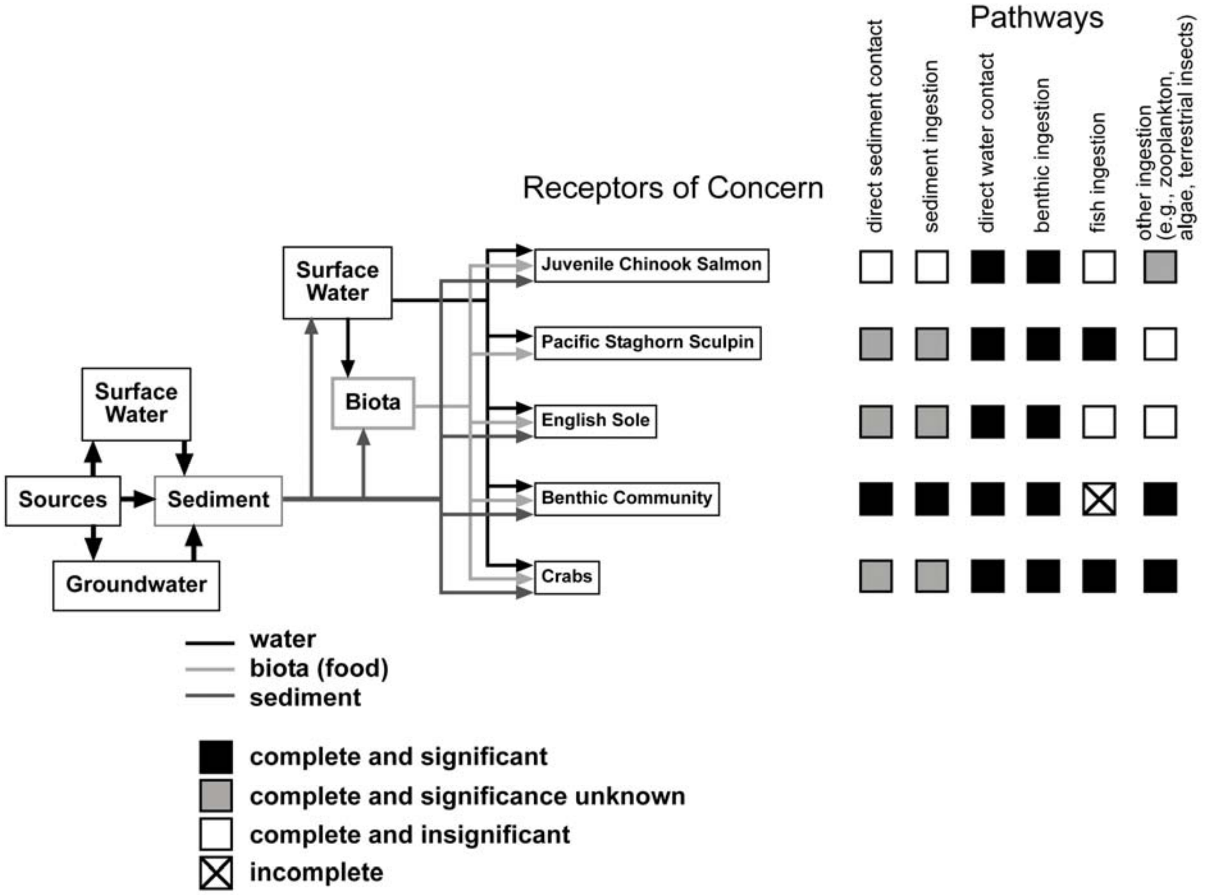
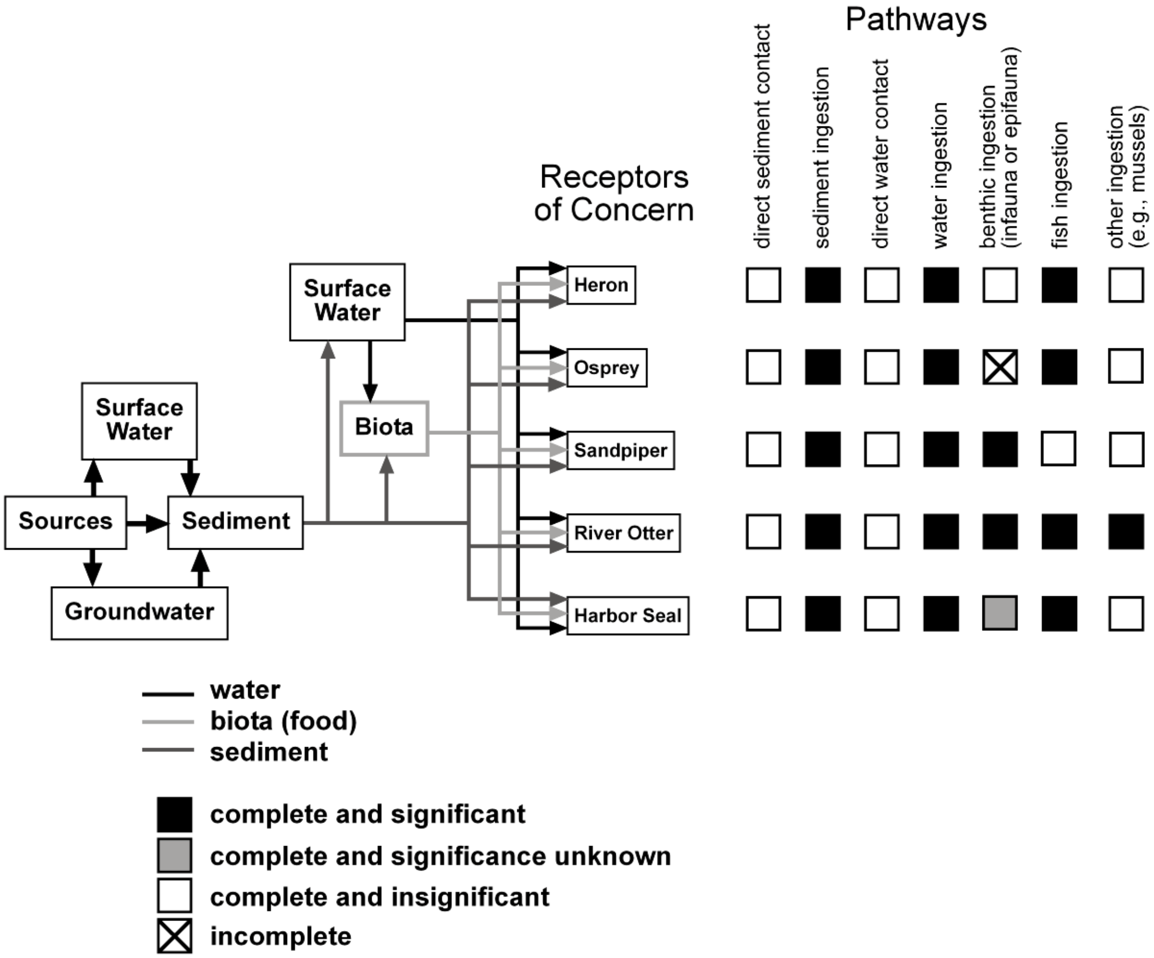
[113] PCBs that accumulate in juvenile salmon during the time they spend in the LDW constitute a small fraction of the PCBs in adult salmon. Most of the PCBs in adults result from foraging activities in Puget Sound or the Pacific Ocean.



Receptors of Concern	direct sediment contact	sediment ingestion	direct water contact	benthic ingestion	fish ingestion	other ingestion (e.g., zooplankton, algae, terrestrial insects)
Juvenile Chinook Salmon	□	□	■	■	□	▨
Pacific Staghorn Sculpin	▨	▨	■	■	■	□
English Sole	▨	▨	■	■	□	□
Benthic Community	■	■	■	■	☒	■
Crabs	▨	▨	■	■	■	■

- ■ complete and significant
- ▨ complete and significance unknown
- □ complete and insignificant
- ☒ incomplete

Figure 5-1. Conceptual site model for fish and the benthic invertebrate community



Receptors of Concern	direct sediment contact	sediment ingestion	direct water contact	water ingestion	benthic ingestion (infauna or epifauna)	fish ingestion	other ingestion (e.g., mussels)
Heron	☐	■	☐	■	☐	■	☐
Osprey	☐	■	☐	■	☒	■	☐
Sandpiper	☐	■	☐	■	■	☐	☐
River Otter	☐	■	☐	■	■	■	■
Harbor Seal	☐	■	☐	■	▨	■	☐

■ complete and significant
▨ complete and significance unknown
☐ complete and insignificant
☒ incomplete

Figure 5-2. Conceptual site model for wildlife

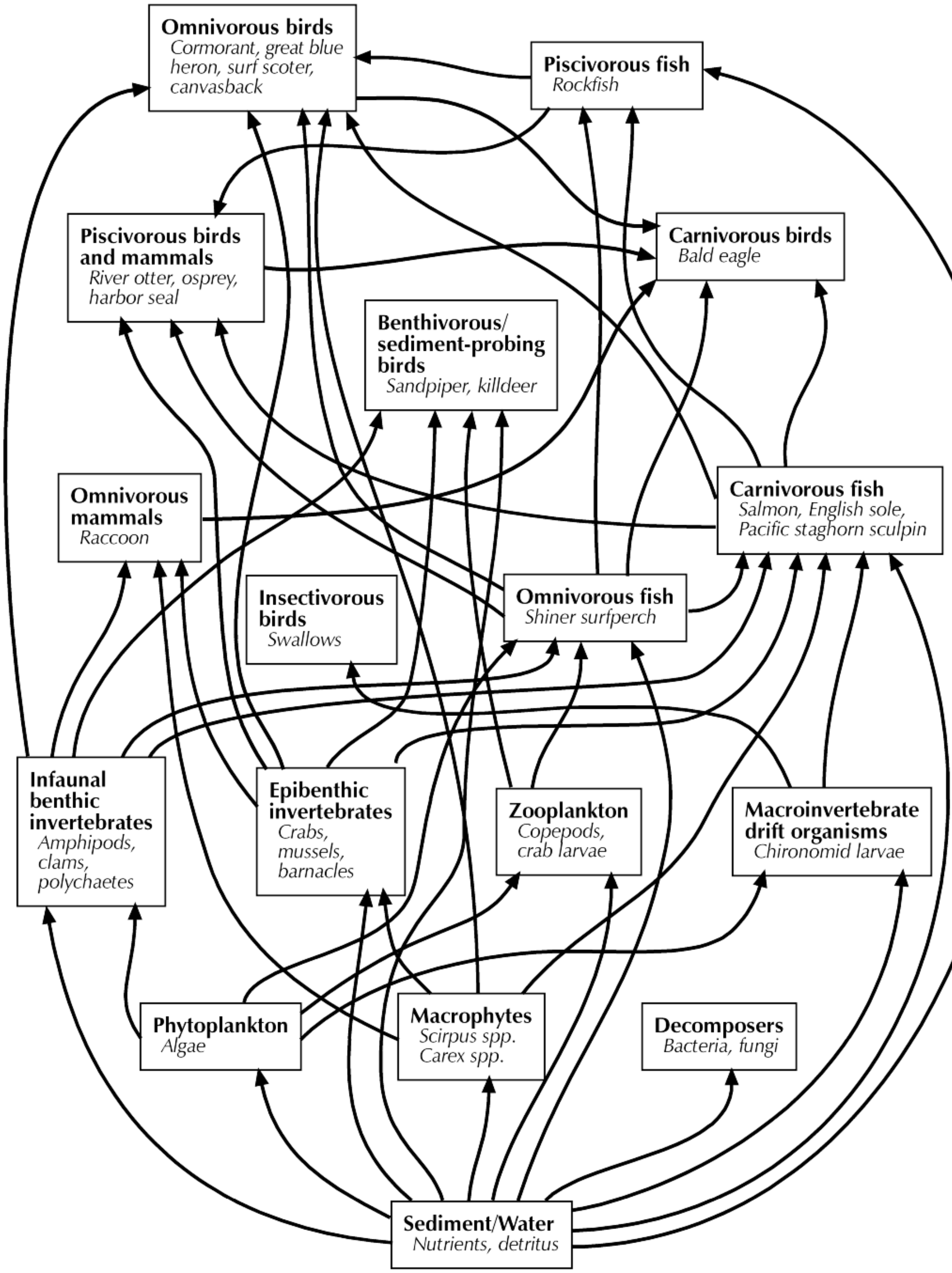


Figure 5-3. Generalized food web diagram for the LDW

Table 5-5. Assessment endpoints for ROCs and measures of effect and exposure

ROC	Assessment Endpoint	Assessment Scale	Measures of Effect	Measures of Exposure
Benthic				
Benthic invertebrate community	survival, growth, reproduction	potential exposure area: small exposure areas for individuals assessment scale: small exposure areas throughout the LDW	SMS and toxicologically based sediment guidelines or TRVs[a]	chemical concentrations in sediment
			water-based TRVs for VOCs	VOC concentrations in porewater
			site-specific toxicity tests	chemical concentrations in sediment samples co-located with toxicity test samples
			tissue-based TRVs for TBT (excluding imposex in gastropods)	TBT concentrations in sediment samples co-located with benthic invertebrate tissue collection
			assessment of imposex in field-collected gastropods	TBT concentrations in sediment samples co-located with gastropod collection
Crabs	survival, growth, reproduction	potential exposure area: crabs may forage throughout the LDW assessment scale: LDW-wide	tissue-based TRVs for decapods	chemical concentrations in crab tissue collected from four tissue sampling areas located throughout the LDW
Fish				
Juvenile chinook salmon	survival and growth	potential exposure area: juvenile salmonids migrate throughout the LDW and forage in shallow areas assessment scale: intertidal areas throughout the LDW	tissue-based TRVs for chemicals evaluated using a critical tissue-residue approach	chemical concentrations in juvenile chinook salmon tissue collected from middle and lower segments of the LDW
			dietary-based TRVs for chemicals evaluated using a dietary approach	chemical concentrations in juvenile chinook salmon prey collected from intertidal habitat throughout the LDW, stomach contents collected from juvenile chinook salmon captured throughout the LDW, and sediment collected from intertidal habitats throughout the LDW
English sole	survival, growth, reproduction	potential exposure area: English sole may forage throughout the LDW assessment scale: LDW-wide	tissue-based TRVs for chemicals evaluated using a critical tissue-residue approach	chemical concentrations in English sole tissue collected from four tissue sampling areas located throughout the LDW
			dietary-based TRVs for chemicals evaluated using a dietary approach	chemical concentrations in English sole prey and sediment collected throughout the LDW
Pacific staghorn sculpin	survival, growth, reproduction	potential exposure area: sculpin may forage throughout the LDW or small segments of LDW assessment scale: LDW-wide and four modeling areas	tissue-based TRVs for chemicals evaluated using a critical tissue-residue approach	chemical concentrations in sculpin tissue collected from four tissue sampling areas located throughout the LDW
			dietary-based TRVs for chemicals evaluated using a dietary approach	chemical concentrations in sculpin prey and sediment collected throughout the LDW and divided into four modeling areas

ROC	Assessment Endpoint	Assessment Scale	Measures of Effect	Measures of Exposure
Wildlife				
Great blue heron	survival, growth, reproduction	potential exposure area: herons may forage in areas of shallow water depths throughout the LDW assessment scale: LDW-wide intertidal	dietary-based TRVs for birds	chemical concentrations in heron prey collected throughout the LDW and in sediment collected from intertidal habitats throughout the LDW
Osprey	survival, growth, reproduction	potential exposure area: osprey may forage from the top meter of water throughout the LDW assessment scale: LDW-wide	dietary-based TRVs for birds	chemical concentrations in osprey prey collected throughout the LDW and in sediment collected from intertidal habitats throughout the LDW
Spotted sandpiper	survival, growth, reproduction	potential exposure area: sandpipers predominantly forage within small home range segments of the LDW assessment scale: three intertidal modeling areas	dietary-based TRVs for birds	chemical concentrations in sandpiper prey and sediment collected from intertidal habitats throughout the LDW
River otter	survival, growth, reproduction	potential exposure area: river otters may forage throughout the LDW assessment scale: LDW-wide	dietary-based TRVs for mammals	chemical concentrations in river otter prey and sediment collected throughout the LDW
Harbor seal	survival, growth, reproduction	potential exposure area: harbor seals may forage throughout the LDW assessment scale: LDW-wide	dietary-based TRVs for mammals	chemical concentrations in harbor seal prey and sediment collected throughout the LDW

[a] A DMMP SL guideline is available for TBT; however, this guideline was not used in this ERA because it is based on an interstitial water concentration. TBT was included as a COPC based on the results of the Phase 1 ERA (Windward 2003b); the potential for adverse effects associated with exposure to TBT was evaluated using benthic invertebrate tissue data consistent with EPA (1999b) and Meador et al. (2002), and through a direct assessment of effects (i.e., imposex in gastropods collected from the LDW).

DMMP – Dredged Material Management Program
COPC – chemical of potential concern
ERA – ecological risk assessment
LDW – Lower Duwamish Waterway
ROC – receptor of concern
SMS – Washington State Sediment Management Standards
SL – screening level
TBT – tributyltin
TRV – toxicity reference value
VOC – volatile organic compound

5.2 BENTHIC INVERTEBRATE ASSESSMENT

This section presents an overview of the risk assessment for the benthic invertebrate community and crabs. The complete assessment is presented in Appendix A, Sections A.3 and A.6.1.

5.2.1 Exposure and effects assessment

This section presents the exposure and effects assessment for benthic invertebrates. The following approaches were used in the assessment for the benthic invertebrate community and crabs:

- Risks to the benthic invertebrate community were evaluated by comparing surface sediment chemical concentrations and site-specific sediment toxicity test results to the SMS; when these were not available, DMMP guidelines and toxicity reference values (TRVs) were used.
- Risks to the benthic invertebrate community from VOC exposure were evaluated by comparing VOC concentrations in porewater[114] to TRVs from the scientific literature.
- Risks to the benthic invertebrate community from TBT exposure were evaluated using results from a site-specific imposex study with gastropods and by comparing TBT concentrations in LDW benthic invertebrate tissue to TRVs from the scientific literature.
- Risks to crabs were evaluated by comparing COPC concentrations in LDW crab tissue to TRVs from the scientific literature.

Each of these approaches is described in the following sections.

5.2.1.1 Sediment data and toxicity test results

The exposure assessment for the benthic invertebrate community summarized the available surface sediment chemistry data for each COPC. The potential for effects on the benthic invertebrate community was assessed by comparing the COPC concentrations in LDW surface sediment to SQS and CSLs of the SMS (WAC 173-204). These numerical chemical criteria are based on apparent effects thresholds (AETs) developed for four different benthic endpoints by the Puget Sound Estuary Program (PSEP) (Barrick et al. 1988). An AET is the highest “no effect” chemical-specific sediment concentration above which a significant adverse biological effect always occurred among the several hundred samples used in its derivation. In general, the lowest of the four AETs for each chemical was identified as the SQS; the second lowest AET was identified as the CSL. According to the SMS (WAC 173-204), locations with all

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[114] Seep water data were not evaluated in the benthic invertebrate risk assessment; however, seep water data and a comparison to Washington State water quality standards are presented in Section 4.2.

chemical concentrations less than or equal to the SQS are defined as having no acute or chronic adverse effects on biological resources, locations with any chemical concentrations between the SQS and CSL are defined as having minor adverse effects, and locations with any chemical concentration greater than the CSL are defined as having more pronounced adverse effects.

Three COPCs identified during the screening process did not have SMS criteria: nickel, total DDTs, and total chlordane. TRVs for nickel were based on toxicologically based DMMP guidelines for nickel. Because the DMMP guidelines for total DDTs and total chlordane were not toxicologically based, TRVs for these COPCs were selected from the scientific literature.

The effects assessment for the benthic invertebrate community also included the results of site-specific toxicity tests conducted using LDW sediment samples during the RI. Three toxicity tests were conducted with surface sediments (0 to 10 cm) collected at each of 48 locations (Windward 2005c, d). Locations were selected for toxicity testing if one or more chemicals exceeded SMS at that location; the suite of locations selected for testing was selected in consultation with EPA and Ecology. The toxicity tests included:

- Acute 10-day amphipod (*Eohaustorius estuarius*) survival test
- Acute 48-hr bivalve larvae (*Mytilus galloprovincialis*) normal survival test
- Chronic 20-day juvenile polychaete (*Neanthes arenaceodentata*) survival and growth test

Biological responses in these tests were evaluated with respect to the biological criteria from the SMS. In addition to the toxicity tests conducted as part of the RI, the effects assessment included results from toxicity tests conducted by King County on sediment samples collected from seven locations at the Duwamish/Diagonal CSO/SD site.

5.2.1.2 VOCs in porewater

Cis-1,2-dichloroethene was the only VOC with a maximum detected concentration in porewater greater than the no-observed-effect concentration (NOEC) in the problem formulation, and thus was the only COPC identified in porewater for the benthic invertebrate community. The concentrations of cis-1,2-dichloroethene detected in porewater at the two sites sampled as part of the RI (GWI and Boeing Plant 2/Jorgensen Forge) were reported in the exposure assessment. The effects assessment identified the NOEC and lowest-observed-effect concentration (LOEC) that were selected for cis-1,2-dichloroethene in water, based on a review of the scientific literature.

5.2.1.3 TBT

The exposure assessment for the benthic invertebrate community for TBT included the results of the co-located benthic invertebrate tissue and sediment samples collected and analyzed as part of the RI from 20 locations throughout the LDW. There was a significant non-linear regression relationship between the TBT concentrations in benthic invertebrate tissue and co-located sediment samples. This relationship was used to estimate the maximum TBT concentration in benthic invertebrates tissue. TBT exposure concentrations included both the maximum concentration estimated from the regression and the maximum detected TBT concentration in benthic invertebrates tissue.

The effects assessment identified the NOAEL and LOAEL for TBT in tissue from the scientific literature. The effects assessment also discussed the results of a site-specific evaluation of imposex (a condition known to be associated with exposure to TBT) in gastropods collected from the LDW. Imposex analysis was performed on multiple individuals of three neogastropod species (*Nassarius mendicus, Astyris gausapata*, and *Olivella baetica)* and on a few individuals (two to five) of four mesogastropod species (*Natica* sp., *Polinices* sp., *L. vincta*, and *Melanella* sp.). Signs of imposex (Stage 2 and maximum relative penis size [RPS] index of 3.4) were observed in *N. mendicus,* but no signs of imposex were observed in the other gastropod species. Stage 2 imposex is not considered to adversely affect the ability of gastropods to reproduce.

5.2.1.4 Crab tissue residue

The exposure assessment for crabs presented UCL concentrations of the two COPCs (zinc and total PCBs) in crab tissue from the LDW. The effects assessment for crab identified NOAELs and LOAELs for zinc and total PCBs in tissue, based on a review of the scientific literature.

5.2.2 Risk characterization and uncertainty analysis

The exposure and effects data for the benthic invertebrate community and for crabs were evaluated in the risk characterization to estimate the potential for adverse effects. The ERA also evaluated uncertainties associated with these risk estimates.

5.2.2.1 Sediment data and toxicity test results

The SQS and CSL were exceeded by 39 chemicals at one or more locations; 2 additional chemicals exceeded the SQS only (Table 5-6). Total DDTs, nickel, and total chlordane also exceeded their DMMP guidelines or literature-based TRVs at one or more locations. Total PCBs and BEHP were the chemicals with the highest number of detected concentrations exceeding both the SQS and CSL criteria.

Table 5-6. Benthic invertebrate community COPCs with detected concentrations in LDW surface sediments greater than SMS criteria, DMMP guidelines, or TRVs

COPC	Number of Detected Concentrations > SQS and ≤ CSL	Number of Detected Concentrations > CSL
Total PCBs	301	173
BEHP	48	58
Mercury	14	23
Lead	2	19

COPC	Number of Detected Concentrations > SQS and ≤ CSL	Number of Detected Concentrations > CSL
Zinc	26	16
Total chlordane[a]	19	14
Copper	0	12
Cadmium	2	11
Silver	0	10
Fluoranthene	31	8
BBP	69	8
Indeno(1,2,3-cd)pyrene	15	8
Chromium	1	8
Arsenic	5	8
Phenol	18	7
Benzo(g,h,i)perylene	9	7
Benzoic acid	0	7
Dibenzo(a,h)anthracene	15	4
Nickel[b]	9	4
Total benzofluoranthenes	5	4
4-Methylphenol	0	4
Phenanthrene	24	3
Total HPAH	21	3
Acenaphthene	16	3
Fluorene	11	3
Benzo(a)anthracene	9	3
Dibenzofuran	7	3
Benzo(a)pyrene	5	3
Total LPAH	3	3
Pyrene	1	3
1,4-Dichlorobenzene	0	3
1,2-Dichlorobenzene	0	3
2-Methylnaphthalene	0	3
Dimethyl phthalate	0	2
Naphthalene	0	2
n-Nitrosodiphenylamine	0	2
Hexachlorobenzene	4	2
Benzyl alcohol	2	2
Chrysene	23	1
Total DDTs[a]	1	1
1,2,4-Trichlorobenzene	0	1

COPC	NUMBER OF DETECTED CONCENTRATIONS > SQS AND ≤ CSL	NUMBER OF DETECTED CONCENTRATIONS > CSL
2,4-Dimethylphenol	0	1
Anthracene	2	0
Pentachlorophenol	1	0

[a] SMS criteria do not exist for these chemicals; the number of exceedances was based on a comparison of sediment chemical concentrations to a TRV.

[b] SMS criteria do not exist for nickel. The DMMP SL and ML values were used for the comparison.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
CSL – cleanup screening level
DMMP – Dredged Material Management Program
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
ML – maximum level
PCB – polychlorinated biphenyl
SL – screening level
SMS – Washington State Sediment Management Standards
SQS – sediment quality standard
TRV – toxicity reference value

The SMS criteria for biological effects were applied to interpret results of site-specific sediment toxicity tests: 18 of the 48 sediment samples (37.5%) did not exceed the SQS biological effects criteria, 11 sediment samples (22.9%) exceeded the SQS biological effects criteria but not the CSL biological effects criteria, and 19 samples (39.6%) exceeded the CSL biological effects criteria. Locations tested and results are shown on Maps 4-14a through 4-14f; results are also presented in Table 4-16. Because these tests are direct measures of effects at a particular location, the uncertainty in their interpretation was considered to be low at the specific locations tested compared with locations with chemistry data but no site-specific toxicity data. The potential for adverse effects was considered to be uncertain at locations where no detected chemicals exceeded the chemical criteria, guidelines, or TRVs but RLs were greater than criteria and guidelines. RLs were greater than criteria for some chemicals because the sample was diluted or because analysis of the chemical posed analytical challenges in the laboratory.

Thiessen polygons were used to estimate the areal extent of potential effects based on combined toxicity test results and surface sediment chemistry data (see Map 4-16). Using this approach, no adverse effects to benthic invertebrates living in intertidal and subtidal sediments were estimated for 75% of the LDW area (i.e., the area in which chemical concentrations were less than or equal to chemical SQS criteria and where sediments were nontoxic according to biological SQS criteria). There was a higher likelihood for adverse effects in approximately 7% of the LDW area, which was designated as having chemical concentrations or biological effects in excess of CSL criteria. The remaining 18% of the LDW area had chemical concentrations or biological effects between the SQS and CSL criteria, indicating that risks to benthic invertebrate communities were considered to be less certain in these areas than in areas with concentrations greater than one or more CSL values. Some uncertainty was associated with these area estimates because areas were estimated by interpolating from individual points at which sediments were sampled.

5.2.2.2 VOCs in porewater

Risks to the benthic invertebrate community from exposure to cis-1,2-dichloroethene were estimated by calculating HQs as the ratio of the concentration in porewater to the NOEC and LOEC concentrations. LOEC-based HQs were less than 1.0 at all locations where cis-1,2-dichloroethene was detected in porewater. At two locations at the GWI site, NOEC-based HQs were greater than 1.0 (4.9 and 21); NOEC-based HQs were less than 1.0 at all other locations. These results indicate it is uncertain whether exposure to cis-1,2-dichloroethene within this small area of the LDW is sufficiently high to result in adverse effects. The limited amount of relevant toxicity data available in the literature was identified as a primary uncertainty in the risk characterization for benthic invertebrates exposed to VOCs in porewater.

5.2.2.3 TBT

Risks to the benthic invertebrate community from exposure to TBT were estimated by calculating HQs as the ratio of the TBT concentration in benthic invertebrate tissue collected from the LDW to the selected NOAEL and LOAEL for benthic invertebrate tissue. Risks to benthic invertebrates from TBT were considered to be very low based on NOAEL-based HQs of less than 1.0. The largest source of uncertainty associated with the critical tissue-residue evaluation of risk to benthic invertebrates was the TRV value. The selected TRV was based on the response of a single species, the polychaete *Armandia brevis,* in a spiked sediment bioassay.

Risks to the benthic invertebrate community from TBT exposure were also considered to be low based on the absence of imposex in all gastropods, except one neogastropod species with imposex characterized as Stage 2, a stage that is not expected to impact reproduction.[115] There was uncertainty in the imposex assessment because: 1) neo- and mesogastropod species may already have been affected by TBT and therefore were no longer present in the LDW, and 2) relatively low numbers of gastropods were collected in the LDW.

5.2.2.4 Crab tissue-residue

Risks to crabs from exposure to total PCBs and zinc were estimated by calculating HQs as the ratio of the COPC concentrations in LDW crab tissue to the selected NOAELs and LOAELs for crab tissue. Total PCB concentrations in crab tissue were equal to the lowest concentrations associated with adverse effects in crabs, indicating the potential for adverse effects (Table 5-7). Zinc concentrations in crab tissue were greater than concentrations associated with no effects but less than those associated with adverse effects, indicating that there is uncertainty as to whether exposure within the LDW

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[115] Imposex is the presence of male reproductive organs in female gastropods, including a penis and vas deferens (a sperm-carrying duct). Imposex was classified according to stages as defined by Oehlmann et al. (1991): Stages 1 through 3 associated with early effects, Stage 4 associated with transitional effects, and Stages 5 and 6 associated with complete sterilization.

would result in adverse effects. The primary uncertainty in the crab risk characterization was the limited number of tissue-based TRVs for total PCBs and zinc available in the literature.

Table 5-7. HQs for crabs using a critical tissue-residue approach

COPC	NOAEL-Based HQ	LOAEL-Based HQ
Total PCBs	**10**	**1.0**
Zinc	**2.5**	0.91

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
Bold identifies NOAEL-based HQs greater than 1.0 or LOAEL-based HQs greater than or equal to 1.0.

Risks to crabs from exposure to total DDTs and methoxychlor were evaluated in the uncertainty section because of the uncertainty associated with JN-qualified tissue organochlorine pesticide data. These data were considered to be highly uncertain because of analytical interference from PCBs, likely resulting in overestimates of organochlorine pesticide concentrations. The LOAEL-based HQs were less than 1.0 for both methoxychlor and total DDTs. The NOAEL-based HQ was 3.6 for methoxychlor and equal to 1.0 for total DDTs. The TRVs were based on a survival endpoint following acute exposures. Overall, risks to crabs from these pesticides are uncertain.

5.3 FISH ASSESSMENT

This section presents an overview of the risk assessment for fish. The complete assessment is presented in Appendix A, Sections A.4 and A.6.2. Three ROCs were selected in the problem formulation to represent fish that use the LDW and may be exposed to sediment-associated chemicals. The three ROCs and COPCs identified for those species were as follows:

- Juvenile chinook salmon—arsenic, cadmium, copper, and vanadium
- English sole—arsenic, cadmium, copper, vanadium, and PCBs
- Pacific staghorn sculpin—arsenic, cadmium, copper, vanadium, TBT, and PCBs

Risks to fish were evaluated using two approaches, depending on the COPC. For TBT and PCBs, concentrations in LDW fish tissue were compared with concentrations in fish tissue associated with adverse effects. For arsenic, cadmium, copper, and vanadium, concentrations in the diet of LDW fish were compared with concentrations associated with adverse effects. A dietary approach was used for these metals because they are metabolically regulated by fish.

5.3.1 Exposure and effects assessment

Estimated concentrations of arsenic, cadmium, copper, and vanadium in fish diets were presented in the exposure assessment for all three fish ROCs. Dietary concentrations of cadmium, copper, and vanadium were estimated using UCL concentrations of COPCs in prey tissue (i.e., benthic invertebrates for all three ROCs plus shiner surfperch for Pacific staghorn sculpin) and incidentally ingested sediment. For arsenic, the linear regression relationship between benthic invertebrate tissue and co-located sediment was used to estimate the arsenic concentration in benthic invertebrate tissue; the UCL of this estimated concentration in tissue was used as the exposure concentration for estimating risk. For juvenile chinook salmon, dietary concentrations were also estimated based on chemical analysis of stomach contents of juvenile chinook salmon collected from the LDW.

To evaluate risk using a tissue-residue approach, the exposure assessment presented the UCL concentration of total PCBs in English sole tissue and total PCBs and TBT in Pacific staghorn sculpin tissue analyzed in whole-body tissue samples collected from the LDW.

For Pacific staghorn sculpin, exposure was evaluated for two foraging scenarios because the foraging range for this species is unknown and may be smaller than the entire LDW. The two scenarios were: 1) foraging throughout the LDW, and 2) foraging in smaller segments of the LDW corresponding to food web modeling areas M1, M2, M3, and M4 (Map 5-1). For English sole, data from the entire LDW was combined to estimate the exposure concentrations.

In the effects assessment, toxicological studies from the scientific literature reporting dietary and fish tissue concentrations associated with potential adverse effects on growth, survival, and reproduction of fish were used to derive NOAELs and LOAELs. As discussed above, studies reporting reproductive endpoints were not used for deriving juvenile chinook salmon TRVs.

5.3.2 Risk characterization and uncertainty analysis

The exposure and effects data were compared in the risk characterization to assess the potential for sediment-associated COPCs to cause adverse effects to juvenile chinook salmon, English sole, and Pacific staghorn sculpin, as discussed in the following subsections.

5.3.2.1 Juvenile chinook salmon

In the risk characterization for juvenile chinook salmon, HQs were calculated as the ratio of the COPC concentrations in the diet (based on either benthic invertebrate data or juvenile chinook stomach content data) to the dietary NOAELs and LOAELs for arsenic, cadmium, copper, and vanadium. Dietary exposure estimates based on benthic invertebrate tissue data from the LDW resulted in higher HQs for all COPCs than did juvenile chinook salmon stomach content data.

No COPC had a LOAEL-based HQ that exceeded 1.0 (Table 5-8). The cadmium LOAEL-based HQs were 1.0 and 0.92 based on concentrations in benthic invertebrates collected from the LDW and in stomach contents, respectively. These HQs are uncertain because the cadmium TRVs were based on juvenile rockfish.

Table 5-8. Summary of risk characterization for juvenile chinook salmon

COPC	Highest NOAEL-Based HQ[a]	Highest LOAEL-Based HQ[a]
Arsenic	**1.1**	0.73
Cadmium	**5.0**	**1.0**
Copper	**1.9**	0.93
Vanadium	**4.0**	0.79

[a] Dietary exposure based on benthic invertebrate tissue data resulted in higher HQs for all COPCs than did juvenile chinook salmon stomach content data. Thus, the HQs presented in this table are based on benthic invertebrate tissue data.

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

NOAEL-based HQs for arsenic, copper, and vanadium were greater than 1.0, whereas LOAEL-based HQs for these COPCs were less than 1.0. Therefore, risks from these COPCs were low and uncertain. The vanadium HQ is based on a NOAEL TRV extrapolated from the only available TRV, which was an unbounded LOAEL. Therefore, risks from vanadium are uncertain and could be higher or lower than the HQs indicate.

An evaluation of risks to juvenile chinook salmon from exposure to endrin was presented in the uncertainty section because of PCB interference in the organochlorine pesticide analyses and resulting uncertainty in the chemistry data. The NOAEL- and LOAEL-based HQs were 3.7 and 0.37, respectively, indicating a low risk with some uncertainty. The HQs are uncertain because: 1) the selected TRVs were based on survival and may have underestimated the potential for sublethal effects, and 2) analytical interference from PCBs was likely to have overestimated actual organochlorine pesticide concentrations in LDW tissue samples.

Risk from chromium was evaluated in the uncertainty section because a LOAEL TRV was not identified. The NOAEL-based HQ was 2.1 based on benthic invertebrate tissue data and 0.18 based on stomach contents data, indicating low risk with some uncertainty.

A number of uncertainties were evaluated in addition to uncertainties in the TRVs discussed above. In the exposure section, there were uncertainties associated with ROC selection, assessment endpoints, foraging habitat, dietary composition, water exposure to PAHs and metals, benthic invertebrate tissue data, and future habitat changes resulting from restoration. Risks to juvenile chinook salmon from exposure to vanadium would be slightly higher if juvenile chinook salmon were assumed to forage throughout the LDW rather than in intertidal areas only. It is not known if juvenile chinook salmon forage in deeper waters, such as the navigation channel, of the LDW, but it is generally considered less likely. In the effects assessment, there were uncertainties associated with exclusion of field studies in TRV selection and interpretation of regional studies.

5.3.2.2 English sole

In the risk characterization for English sole, HQs were calculated as the ratio of the COPC concentrations in the diet to the dietary NOAELs and LOAELs for arsenic, cadmium, copper, and vanadium. For the critical tissue-residue approach, HQs were calculated as the ratio of the total PCB concentration in English sole tissue to the tissue-based NOAEL and LOAEL TRVs.

Dietary concentrations for two of the six COPCs for fish (cadmium and vanadium) were greater than concentrations associated with adverse effects for English sole (LOAEL-based HQs for cadmium and vanadium were both 1.2); therefore, there is a potential for adverse effects (Table 5-9). Effects data for cadmium were highly variable, and there was uncertainty associated with the selected TRV because effects were partially attributed to reduced feeding. Thus, risks from cadmium were low but uncertain. The vanadium LOAEL-based HQ was based on an unbounded LOAEL reported in the only dietary toxicity study identified. The NOAEL TRV was estimated from the unbounded LOAEL TRV using a safety factor of 5. Because of the very limited effects data, risk from vanadium was uncertain and could have been higher or lower than the HQs indicated.

Table 5-9. Summary of risk characterization for English sole

APPROACH	COPC	NOAEL-BASED HQ	LOAEL-BASED HQ
Tissue residue	total PCBs	**4.9 – 25**[a]	0.98 – **5.0**[a]
Dietary	arsenic	**1.2**	0.80
	cadmium	**6.1**	**1.2**
	copper	**1.9**	0.93
	vanadium	**5.9**	**1.2**

[a] Because of uncertainty in the LOAEL, LOAEL-based HQs were calculated from a range of effects concentrations reported in Hugla and Thome (1999). The NOAEL TRV range was estimated by dividing the LOAEL TRV range by an uncertainty factor of 5.

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

For PCBs, the LOAEL-based HQs ranged from 0.98 to 5.0 based on toxicity information presented in one study reporting the lowest TRVs (Hugla and Thome 1999). A NOAEL was not available from this study, and no NOAELs were reported for any other study below the selected LOAEL range. There is uncertainty in the risk estimates because of uncertainties with the study reporting the lowest TRVs. If the study reporting the next higher LOAEL TRV had been selected, exposures would have been below the effects thresholds (LOAEL-based HQ of 0.28). Risks were also estimated from PCB congener concentrations in English sole tissue using a TEQ approach; those risks were determined to be very low. TEQs were calculated using only the dioxin-like PCB congeners because dioxin and furan tissue data were not available. Thus, exposure associated with dioxin-like chemicals was likely underestimated, although it is uncertain whether the inclusion of dioxins and furan exposure would have changed the risk conclusions.

Estimated dietary exposures of English sole to two additional COPCs (arsenic and copper) were greater than their respective no-effects levels but lower than the adverse effect levels associated with survival, growth, or reproduction, indicating that there is uncertainty as to whether exposure to these metals within the LDW is sufficiently high to result in adverse effects.

An evaluation of risks to English sole from exposure to endosulfan and endrin was presented in the uncertainty section because of PCB interference in the organochlorine pesticide analyses. The LOAEL-based HQs were less than 1.0 for endosulfan and 1.2 for endrin. These HQs are uncertain because: 1) the selected TRVs were based on survival and may have underestimated the potential for sublethal effects, and 2) analytical interference from PCBs is likely to have overestimated actual organochlorine pesticide concentrations in LDW tissue samples.

LOAEL TRVs were not available for chromium and benzoic acid, so risks from these chemicals were evaluated in the uncertainty analysis. Comparison of dietary exposure concentrations to NOAEL TRVs resulted in NOAEL-based HQs of 1.1 and 1.5, respectively, for chromium and benzoic acid, indicating low risk with some uncertainty.

Regional studies, which included samples collected from the LDW, indicated an increased risk of adverse effects on English sole reproduction in the LDW. Potential causal factors included PAHs and PCBs; however, these effects cannot be conclusively associated with exposure to specific chemicals or chemical mixtures in the LDW because, among other factors, of the complex mixtures of chemicals in the field and uncertainties in the English sole home range.

A number of uncertainties were evaluated in addition to uncertainties in the TRVs and additional chemicals discussed above. Uncertainties associated with ROC selection, incidental sediment ingestion, dietary composition, foraging range, water exposure to PAHs and metals, benthic invertebrate tissue data, and future habitat changes resulting from restoration were not expected to have an effect on risk conclusions. Uncertainties associated with the use of field studies were also discussed.

5.3.2.3 Pacific staghorn sculpin

In the risk characterization for Pacific staghorn sculpin, HQs were calculated as the ratio of the COPC concentrations in the diet to the dietary NOAELs and LOAELs for arsenic, cadmium, copper, and vanadium. For the critical tissue-residue approach, HQs were calculated as the ratio of the total PCB or TBT concentrations in Pacific staghorn sculpin tissue to the tissue-based NOAEL and LOAEL TRVs.

Both the NOAEL- and LOAEL-based HQs were greater than 1.0 for vanadium in modeling areas M2 and M3 (NOAEL- and LOAEL-based HQs of 1.2 and 5.9, respectively, in both areas) (Table 5-10). In modeling areas M1 and M4 and LDW-wide, NOAEL-based HQs were greater than 1.0, but LOAEL-based HQs were less than or equal to 1.0. As discussed above for English sole, because of the limited amount of effects data available, risks from vanadium were uncertain and could have been higher or lower than the HQs indicated.

Table 5-10. Summary of risk characterization for Pacific staghorn sculpin

Approach	**COPC**	**Foraging Assumption**	**NOAEL-Based HQ or Range of HQs**[a]	**LOAEL-Based HQ or Range of HQs**[a]
Tissue residue	total PCBs	LDW-wide	**2 – 11**[b]	0.42 – **2.1**[b]
		modeling areas	**1.5** – **19**	0.30 – **3.8**
	TBT	LDW-wide	**2.0**	0.23
		modeling areas	**1.6 – 2.9**	0.18 – 0.33
Dietary	arsenic	LDW-wide	0.75	0.50
		modeling areas	0.60 – 0.80	0.40 – 0.53
	cadmium	LDW-wide	**3.8**	0.76
		modeling areas	**3.0 – 5.2**	0.60 – **1.0**
	copper	LDW-wide	**1.1**	0.56
		modeling areas	0.90 – **1.5**	0.45 – 0.77
	vanadium	LDW-wide	**4.0**	0.79
		modeling areas	**3.2 – 5.9**	0.65 – **1.2**

[a] Range of HQs for the four modeling areas evaluated for Pacific staghorn sculpin.

[b] Because of uncertainty in the LOAEL, LOAEL-based HQs were calculated from a range of effects concentrations reported in Hugla and Thome (1999). The NOAEL TRV range was estimated by dividing the LOAEL TRV range by an uncertainty factor of 5.

COPC – chemical of potential concern
HQ – hazard quotient
LDW – Lower Duwamish Waterway
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

For PCBs, the LOAEL-based HQs ranged from 0.30 to 3.8 based on toxicity information presented in one study reporting the lowest TRVs (Hugla and Thome 1999) and exposure data from discrete areas in the LDW as well as LDW-wide. A NOAEL was not available from Hugla and Thome (1999), and no NOAELs below the selected LOAEL range were reported for any other study. There is uncertainty in the risk estimates because of uncertainties with the study reporting the lowest TRVs. If the study reporting the next higher LOAEL TRV had been selected, the LOAEL-based HQ would have been 0.12 for LDW-wide exposure instead of ranging from 0.42 to 2.1.

The LOAEL-based HQ for cadmium was equal to 1.0 in modeling area M3 but lower than 1.0 in all other modeling areas and LDW-wide. As discussed above for English sole, there was some uncertainty associated with the selected cadmium TRV because effects were partially attributed to reduced feeding. In addition, data used to represent crabs in Pacific staghorn sculpin dietary exposure calculations were uncertain; the LOAEL-based HQ for cadmium would have been 1.9 if crab data had been used instead of benthic invertebrate data. LOAEL-based HQs were less than 1.0 for TBT and copper. NOAEL-based HQs for TBT and copper were greater than 1.0 LDW-wide and in one or more modeling areas, indicating that the potential for adverse effects was low but uncertain.

An evaluation of risks to Pacific staghorn sculpin from exposure to endosulfan and endrin was presented in the uncertainty section because the presence of PCBs interfered in the organochlorine pesticide analyses. This evaluation indicated a low risk from endosulfan and the potential for adverse effects from endrin (LOAEL-based HQ of 3.1). These HQs were uncertain because: 1) the selected TRVs were based on survival and may have underestimated the potential for sublethal effects, and 2) analytical interference from PCBs is likely to have overestimated actual organochlorine pesticide concentrations in LDW tissue samples.

LOAEL TRVs were not available for chromium and benzoic acid, so risks from these chemicals were evaluated in the uncertainty analysis. Comparison of dietary exposure concentrations to NOAEL TRVs resulted in NOAEL-based HQs of 0.85 and 2.1, respectively, for chromium and benzoic acid, indicating low risk with some uncertainty.

A number of uncertainties were evaluated in addition to uncertainties in the TRVs discussed above. Uncertainties associated with the COPC screen, ROC selection, water exposure to PAHs and metals, PCB analysis approach (total PCBs versus PCB TEQ), and future habitat changes resulting from restoration were not expected to have an effect on risk conclusions.

In the uncertainty analysis, uncertainty associated with the benthic invertebrate portion of Pacific staghorn sculpin diet was evaluated. Use of large crab tissue data versus benthic invertebrate tissue samples (which included crabs and shrimps < 20 mm) resulted in slight changes in LOAEL-based HQs for cadmium, copper, and vanadium. For arsenic, no HQs changed from less than 1.0 to greater than 1.0 or vice versa, based on this data substitution.

5.4 WILDLIFE ASSESSMENT

This section presents an overview of the risk assessment for wildlife. The complete assessment is presented in Appendix A, Sections A.5 and A.6.3. Five ROCs were selected in the problem formulation to represent wildlife that use the LDW and may be exposed to sediment-associated chemicals. The five ROCs and associated COPCs were:

- Spotted sandpiper – arsenic, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, vanadium, zinc, and PCBs
- Great blue heron – chromium, lead, mercury, and PCBs
- Osprey – arsenic, chromium, lead, mercury, and PCBs
- River otter – arsenic, cobalt, mercury, selenium, and PCBs
- Harbor seal – mercury and PCBs

Risks to wildlife ROCs were evaluated by comparing estimated COPC concentrations as a dietary dose for each ROC to dietary doses associated with adverse effects from the scientific literature.

5.4.1 Exposure and effects assessment

For wildlife ROCs, the daily exposure doses of COPCs through the ingestion of prey, water, and incidental sediment were estimated in the exposure assessment. Exposure doses were calculated as follows:

$$\text{Exposure dose} = \frac{[(\text{FIR} \times C_{\text{food}}) + (\text{WIR} \times C_{\text{water}}) + (\text{SIR} \times C_{\text{sed}})] \times \text{SUF}}{\text{BW}} \qquad \textbf{Equation 5-1}$$

Where:

Exposure dose = COPCs ingested per day via food, water, and sediment (mg COPC/kg body weight/day)
FIR = food ingestion rate (kg food dw/day)
C_{food} = concentration in prey items (mg COPC/kg food dw)
WIR = water ingestion rate (L water/day)
C_{water} = concentration in water (mg COPC/L water)
SIR = sediment ingestion rate (kg sediment dw/day)
C_{sed} = concentration in sediment (mg COPC/kg dw)
SUF = site use factor (unitless); fraction of time that a receptor spends foraging in the LDW relative to the entire home range
BW = ROC species body weight (kg ww)

Body weights, and food, water, and sediment ingestion rates for each receptor were derived from the literature. Site use factors for great blue heron, osprey, and harbor seal were less than 1.0 because site-specific data showed that in addition to foraging in the LDW, these receptors also foraged outside the LDW. Spotted sandpiper and river otter were assumed to forage only in the LDW. Within the LDW, the foraging areas for all wildlife ROCs except spotted sandpiper, were assumed to be the entire LDW. Spotted sandpiper foraging areas were identified based on a field survey conducted as part of the RI in 2004 (Windward 2004j). The field survey identified potential nesting and foraging habitat. Assuming a foraging range of about 1 mile from the nesting habitat, three hypothetical exposure areas were created. Within each area, two foraging scenarios were evaluated: one in which spotted sandpipers forage in only high-quality habitat, and another in which they forage in both high- and poor-quality habitats. These two foraging scenarios in each of the three exposure areas resulted in a total of six exposure scenarios for spotted sandpiper (Map 5-2).

Concentrations of COPCs in prey, sediment, and water were calculated as the UCL of the data within the relevant exposure area for each ROC, with a few exceptions. Total PCBs in sediment were calculated as the 95% UCL of the SWAC within relevant exposure areas. A linear regression relationship between benthic invertebrate tissue and co-located sediment was used to estimate UCL on the mean benthic invertebrate tissue concentrations of arsenic and total PCBs in each exposure area.

The effects assessment presented a review of the scientific literature for dietary exposure doses associated with adverse effects on survival, growth, and reproduction in birds and mammals. NOAELs and LOAELs were selected for each COPC based on these toxicity data.

5.4.2 Risk characterization and uncertainty analysis

The exposure and effects data were compared in the risk characterization to assess the potential for sediment-associated COPCs to have adverse effects on the ROCs. For each ROC, HQs were calculated as the ratio of the COPC concentration in the daily exposure dose to the NOAELs and LOAELs. The following sections present the risk conclusions for each wildlife receptor.

5.4.2.1 Spotted sandpiper

The following COPCs had LOAEL-based HQs greater than or equal to 1.0 for spotted sandpiper: chromium, copper, lead, mercury, and vanadium (Table 5-11). LOAEL-based HQs ranged from 1.0 to 5.5 for these COPCs. LOAEL-based HQs were equal to or greater than 1.0 in Exposure Area 2 for chromium (1.8) and lead (5.5), in Exposure Area 3 for copper (1.1) and mercury (1.0), and in all three exposure areas for vanadium (1.0 to 1.4) (see Map 5-2 for exposure areas). Risks to spotted sandpiper from copper and vanadium may be under- or overestimated because the selected TRVs were based on subchronic growth endpoints. The ingested dose was primarily from sediment for copper and lead in Exposure Area 3 and for vanadium in all areas. Bioavailability of metals in sediment is unlikely to be 100%, so the LOAEL-based HQs (ranging from 1.0 to 1.5) for copper, lead, and vanadium may have been overestimated. Overall, these findings indicated risks to spotted sandpiper in some areas of the LDW from exposure to chromium, copper, lead, mercury, and vanadium, but these risks are expected to be low.

Table 5-11. Summary of risk characterization for spotted sandpiper

COPC	RANGE OF NOAEL-BASED HQs[a]	RANGE OF LOAEL-BASED HQs[a]
Arsenic	0.34 – 0.58	< 0.1 – 0.15
Cadmium	< 0.1 – 0.21	< 0.1
Chromium	**1.3** – **8.8**	0.26 – **1.8**
Cobalt	0.22 – 0.27	< 0.1
Copper	0.62 – **1.5**	0.45 – **1.1**
Lead	0.58 – **19**	0.17 – **5.5**
Mercury	**1.1** – **5.3**	0.21 – **1.0**
Nickel	< 0.1 – 0.22	< 0.1– 0.11
Selenium	0.62 – 0.90	0.38 – 0.55
Vanadium	**2.0** – **2.7**	**1.0** – **1.4**
Zinc	0.41 – 0.78	0.27 – 0.52
Total PCBs	0.51 – **2.0**	0.18 – 0.71
PCB TEQs	**1.9** – **10**	0.18 – **1.5**

[a] Range of HQs for the six spotted sandpiper exposure scenarios for each COPC.

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

For total PCBs, the NOAEL-based HQs were greater than 1.0 in some of the spotted sandpiper exposure areas, but LOAEL-based HQs in all areas were less than 1.0, indicating low risk with some uncertainty because the true threshold of effects between the NOAEL and the LOAEL is not known. For PCB TEQs, the LOAEL-based HQs were less than 1.0, except in Exposure Area 2 under the high-quality foraging habitat scenario, in which the LOAEL-based HQ was 1.5. Uncertainty was considered to be high for PCB TEQ risk estimates because the TRVs were based on a study that used an acute weekly dose via intraperitoneal (IP) injection, and TEFs derived from studies of toxicity to eggs were applied to dietary exposure estimates for adults. Therefore, risks to spotted sandpiper from PCBs were considered to be low in most areas of the LDW, with some uncertainty.

For the remaining COPCs (arsenic, cadmium, cobalt, nickel, selenium, and zinc), all NOAEL- and LOAEL-based HQs were less than 1.0 in all sandpiper exposure areas. Exposure of spotted sandpipers to these COPCs was not expected to result in adverse effects.

An evaluation of risks from exposure to total DDT was conducted in the uncertainty section because of PCB interference in the organochlorine pesticide analyses. The NOAEL-based HQs were greater than 1.0 in some of the spotted sandpiper exposure areas, but LOAEL-based HQs were less than 1.0, indicating low risk. It is likely that HQs would have been lower if there had been no analytical interferences from PCBs because the interference is known to bias the DDT concentrations high. Therefore, risks to spotted sandpiper from DDT are expected to be very low in most areas of the LDW, with some uncertainty.

A number of additional uncertainties were evaluated in the uncertainty section. Uncertainties in COPC selection, food ingestion rate, direct sediment contact, incidental sediment ingestion, dietary composition, and site use were expected to have minimal or no effect on risk conclusions. Risks were considered to be uncertain for cadmium, cobalt, copper, nickel, vanadium, and zinc because TRVs were based on subchronic exposure and growth endpoints. Uncertainties in toxicity data for chromium, lead, mercury, selenium, and PCBs (based on a total PCB approach) could also have affected risk estimates. Risks could have been underestimated for arsenic because of the lack of arsenite toxicity data.

5.4.2.2 Great blue heron

The NOAEL- and LOAEL-based HQs for great blue heron were all less than 1.0, indicating very low risks to great blue herons were expected from exposure to chromium, lead, mercury, and PCBs in the LDW (Table 5-12). For chromium and for PCB TEQs, there was high uncertainty in the TRVs, but the LOAEL TRVs would have needed to be lower by a factor of 14 for the LOAEL-based HQs to equal 1.0. There was also some uncertainty in the PCB TEQ risk estimate because TEFs derived from studies of toxicity to eggs were applied to dietary exposure estimates for adults.

Table 5-12. Summary of risk characterization for great blue heron

COPC	NOAEL-BASED HQ	LOAEL-BASED HQ
Chromium	< 0.1	< 0.1
Lead	< 0.1	< 0.1
Mercury	0.17	< 0.1
Total PCBs	0.35	0.12
PCB TEQs	0.66	< 0.1

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Uncertainties in incidental sediment ingestion, dietary composition, and site use were evaluated and expected to have minimal or no effect on risk conclusions. The uncertainty analysis section also evaluated risks from the exposure of great blue heron to total DDTs. The NOAEL-based HQ was less than 1.0, indicating very low risk.

5.4.2.3 Osprey

The NOAEL- and LOAEL-based HQs for osprey for chromium, lead, and mercury were less than 1.0, indicating that there are very low risks to osprey from exposure to these COPCs in the LDW (Table 5-13). There was high uncertainty in the chromium TRV, but the LOAEL TRV would have needed to be lower by a factor of 50 for the LOAEL-based HQ to equal 1.0.

Table 5-13. Summary of risk characterization for osprey

COPC	NOAEL-Based HQ	LOAEL-Based HQ
Chromium	0.11	< 0.1
Lead	< 0.1	< 0.1
Mercury	0.28	< 0.1
Total PCBs	0.65	0.23
PCB TEQs	**1.6**	0.16

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
Bold identifies NOAEL-based HQs greater than 1.0.

NOAEL- and LOAEL-based HQs for total PCBs were less than 1.0. The LOAEL-based HQ for PCBs using the PCB TEQ approach was less than 1.0, but the NOAEL-based HQ was 1.6, indicating a low risk with uncertainty. There were no PCB congener data for juvenile chinook salmon, one of the three species assumed to be ingested by osprey in this assessment, so the portion of prey ingestion that had been assigned to juvenile chinook salmon (45%) was divided proportionally among shiner surfperch and English sole. Shiner surfperch had higher PCB concentrations than did juvenile chinook salmon, so is it possible that PCB TEQ HQs would have been lower if PCB congener data had been available for juvenile salmon. There was also high uncertainty in the PCB TEQ TRV because: 1) birds in the study were given an acute weekly dose via IP injection rather than via the diet, which may result in an overestimate of exposure and resulting toxicity, and 2) TEFs derived from studies of toxicity to eggs were applied to dietary exposure estimates for adults.

The uncertainty assessment included an estimate of total PCB concentrations in osprey eggs using a biomagnification factor relating total PCB concentrations in fish to concentrations in bird eggs. The comparison of the estimated total PCB concentration in eggs to the LOAEL TRV resulted in an HQ of 1.4. This approach was highly uncertain, resulting in potential over- or underestimates of risk. Thus, for total PCBs, the ingested dose approach with its associated uncertainties indicated that risks to osprey were low, while risks calculated using estimated concentrations in eggs indicated that risks were low with a high level of uncertainty.

Uncertainties in incidental sediment ingestion and dietary composition were evaluated and expected to have minimal or no effect on risk conclusions. The uncertainty section also evaluated risks from exposure of osprey to total DDTs. The NOAEL-based HQ was less than 1.0, indicating very low risk.

5.4.2.4 River otter

The NOAEL- and LOAEL-based HQs for PCBs of 5.8 and 2.9, respectively, using the total PCB approach, indicated a risk to river otters from PCB exposure (Table 5-14). The PCB TRVs were based on an 18-month study with mink in which reduced growth of offspring was reported. The NOAEL-based HQ of 4.5 using PCB congener data and the PCB TEQ approach indicated that there could be some risk to river otters from PCB exposure, although the LOAEL-based HQ was less than 1.0 using this approach. The PCB TEQ TRVs were based on a 90-day study with guinea pigs in which reduced body weights were reported. There were some uncertainties associated with the PCB TRVs and with the TEF approach, but it was not known whether these uncertainties would result in an under- or overestimate of risk. Therefore, risks to river otter from PCBs were considered to be low, with some uncertainty.

Table 5-14. Summary of risk characterization for river otter

COPC	NOAEL-BASED HQ	LOAEL-BASED HQ
Arsenic	0.12	< 0.1
Cobalt	0.16	< 0.1
Mercury	**2.8**	0.57
Selenium	0.58	0.40
Total PCBs	**5.8**	**2.9**
PCB TEQs	**4.5**	0.59

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

For mercury, the LOAEL-based HQ was less than 1.0 (0.57), but the NOAEL-based HQ was greater than 1.0 (2.8). There was some uncertainty in the TRV, which was based on a chronic study of rats resulting in effects on growth, because the NOAEL was estimated from the LOAEL using an uncertainty factor of 5. Therefore, risks to river otter from mercury were considered to be low, with some uncertainty.

The NOAEL- and LOAEL-based HQs for river otter for arsenic, cobalt, and selenium were less than 1.0, indicating very low risks to river otters from exposure to these COPCs in the LDW.

Uncertainties in food and incidental sediment ingestion rates and dietary composition were evaluated and were expected to have minimal or no effect on risk conclusions.

5.4.2.5 Harbor seal

The NOAEL- and LOAEL-based HQs for harbor seal for mercury and PCBs were less than 1.0, indicating very low risk to harbor seals from exposure to these COPCs in the LDW (Table 5-15). Uncertainties in incidental sediment ingestion rates and dietary composition were expected to have minimal or no effect on risk conclusions.

Table 5-15. Summary of risk characterization for harbor seal

COPC	NOAEL-BASED HQ	LOAEL-BASED HQ
Mercury	0.19	< 0.1
Total PCBs	0.44	0.22
PCB TEQs	0.35	< 0.1

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

5.5 SUMMARY AND SELECTION OF ECOLOGICAL RISK DRIVERS

In summary, risk estimates for PCBs indicated a potential for adverse effects to the benthic invertebrate community, crabs, spotted sandpiper, and river otter (Table 5-16), and thus, PCBs were identified as a COC. COCs were defined as chemicals with LOAEL-based HQs greater than or equal to 1.0, which indicates a potential for adverse effects. There is a potential for adverse effects to English sole, and Pacific staghorn sculpin; however, the risk estimates for PCBs for these two fish species are uncertain both because the exposure concentrations were in between the concentrations selected as the LOAEL TRV range and because of the uncertainty in the study that served as the basis for the LOAEL TRV range itself. There is also a potential for adverse effects for osprey from PCBs based on the PCB TEQ, which was highly uncertain. Other COCs with exposures greater than or equal to levels associated with adverse effects for at least one fish or wildlife receptor were cadmium, chromium, copper, lead, mercury, and vanadium. Numerous additional chemicals pose a risk to the benthic invertebrate community, as indicated in Table 5-6. No quantitative ecological risk estimates were calculated for dioxins and furans.

Table 5-16. List of chemicals with NOAEL- or LOAEL-based HQs ≥ 1.0

CHEMICAL	ROC	NOAEL-BASED HQ	LOAEL-BASED HQ
COCs with LOAEL-based HQs ≥ 1.0[a]			
Total PCBs	English sole	**4.9 – 25[b]**	**0.98 – 5.0[b]**
	Pacific staghorn sculpin	**3.8 – 19[b]**	**0.76 – 3.8[b]**
	crabs	**10**	**1.0**
	river otter	**5.8**	**2.9**
PCB TEQs	spotted sandpiper	**1.9 – 15**	0.18 – **1.5**
Cadmium	juvenile chinook salmon	**5.0**	**1.0**
	English sole	**6.1**	**1.2**
	Pacific staghorn sculpin	**3.0 – 5.2**	0.60 – **1.0**
Chromium	spotted sandpiper	**1.3 – 8.8**	0.26 – **1.8**
Copper	spotted sandpiper	0.62 – **1.5**	0.45 – **1.1`**
Lead	spotted sandpiper	0.58 – **19**	0.17 – **5.5**
Mercury	spotted sandpiper	**1.1 – 5.3**	0.21 – **1.0**
Vanadium	English sole	**5.9**	**1.2**
	Pacific staghorn sculpin	**3.2 – 5.9**	0.65 **– 1.2**
	spotted sandpiper	**2.0 – 2.7**	**1.0 – 1.4**
COPCs with NOAEL-based HQs ≥ 1.0 and LOAEL-based HQs < 1.0[c]			
Total PCBs	English sole	**1.4**	0.28
	Pacific staghorn sculpin	0.42 – **1.1**	< 0.1 – 0.22
	spotted sandpiper	0.51 – **2.0**	0.18 – 0.71
PCB TEQs	osprey	**1.6**	0.16
	river otter	**4.5**	0.59
Arsenic	juvenile chinook salmon	**1.1**	0.73
	English sole	**1.2**	0.80
	crabs	**3.9**	na
Benzoic acid	English sole	**1.5**	na
	Pacific staghorn sculpin	**2.1**	na
Cadmium	Pacific staghorn sculpin	**3.0 – 4.9**	0.60 – 0.98
Chromium	juvenile chinook salmon	**2.1**	na
	English sole	**1.1**	na

Chemical	ROC	NOAEL-Based HQ	LOAEL-Based HQ
Copper	juvenile chinook salmon	**1.9**	0.93
	English sole	**1.9**	0.93
	Pacific staghorn sculpin	0.9 – **1.5**	0.45 – 0.77
Mercury	river otter	**2.8**	0.57
TBT	Pacific staghorn sculpin	**1.6** – **2.9**	0.18 – 0.33
Vanadium	juvenile chinook salmon	**4.0**	0.79
Zinc	crabs	**2.5**	0.91

Note: HQs reported for fish are the highest HQs in cases where more than one approach was used.

[a] The LOAEL-based HQs for endrin were 1.2 and 3.1 for English sole and Pacific staghorn sculpin, respectively, based on risk calculations discussed in the uncertainty analysis. These calculations were discussed only in the uncertainty analysis because of analytical interferences from PCB Aroclors in the organochlorine pesticide analyses, resulting in uncertainties in pesticide identification and a high bias in pesticide concentrations.

[b] LOAEL-based HQs were calculated from a range of effects concentrations reported in Hugla and Thome (1999) because of uncertainty in the LOAEL. The NOAEL TRV range was estimated by dividing the LOAEL TRV range by an uncertainty factor of 5. Ranges reported for Pacific staghorn sculpin also included the range in exposure estimates for areas smaller than the entire LDW.

[c] The NOAEL-based HQs were greater than 1.0 for the following COPC/ROC pairs based on risk calculations discussed in the uncertainty analysis: 1) total DDTs and spotted sandpiper (2.6 to 4.3), 2) endrin and juvenile chinook salmon (3.6), 3) alpha-endosulfan and English sole (6.8) and Pacific staghorn sculpin (2.3), 4) beta-endosulfan and English sole (29) and Pacific staghorn sculpin (6.6), 5) endrin and juvenile chinook salmon (3.6), and 6) methoxychlor and crabs (3.6). These calculations were discussed in the uncertainty analysis because of analytical interferences from PCB Aroclors in the organochlorine pesticide analyses, resulting in uncertainties in pesticide identification and a high bias in pesticide concentrations.

COC – chemical of concern
COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
ROC – receptor of concern
TBT – tributyltin
TEQ – toxic equivalent

Bold identifies NOAEL-based HQs greater than 1.0 or LOAEL-based HQs greater than or equal to 1.0.

Chemicals were identified as risk drivers for ecological receptors based on the risk estimates, uncertainties discussed in the ERA (Appendix A), preliminary natural background concentrations, and residual risks following planned early actions in the LDW. The risk drivers from both the ERA and the HHRA will be the focus of remedial analyses in the FS.

In consultation with EPA and Ecology, PCBs were identified as a risk driver for river otter because estimated exposures of river otter were greater than the LOAEL by a factor of 2.9 and uncertainties in the risk estimate were relatively low. In addition, 41 chemicals were selected as risk drivers for benthic invertebrates because detected concentrations of these 41 chemicals exceeded the SQS of the Washington SMS in one or more locations.

Other COCs that exceeded risk thresholds (LOAEL-based HQ greater than or equal to 1.0) were not selected as risk drivers because of high uncertainty in the effects or exposure data, comparisons to preliminary background concentrations, or the expectation of low residual risk following remediation in EAAs, as discussed in detail in Section 7 of the ERA. COCs that were not selected as risk drivers will be addressed through focused evaluation in the FS. These chemicals may also be considered in remedial design for specific areas in or near the LDW and in the post-remedial monitoring program that is part of the 5-year review that EPA conducts at all Superfund sites where hazardous substances, pollutants, or contaminants remain in place after cleanup is completed.

6 Summary of the Baseline Human Health Risk Assessment

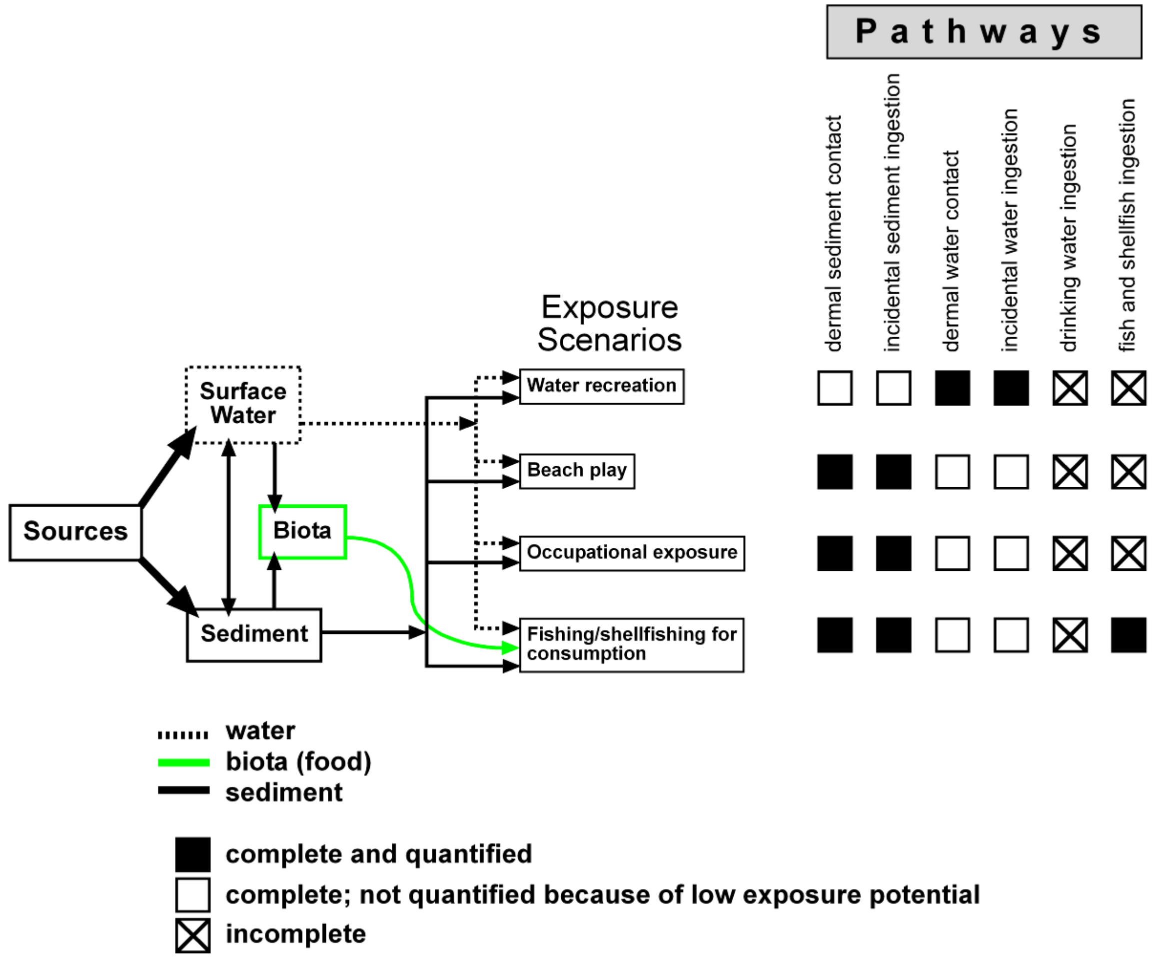
The baseline HHRA estimated risks for people who may be exposed to COPCs present in sediment, water, and aquatic biota in the LDW. The dataset used in the baseline HHRA consisted of historical data used in the Phase 1 HHRA and sediment and tissue chemistry data collected from the LDW as part of the RI, as summarized in Appendix B, Section B.2.1. This section summarizes the exposure assessment (including the conceptual site model and selection of exposure pathways, the screening and evaluation of COPCs, and the selection of exposure parameters), toxicity assessment, risk characterization, uncertainty analysis, and selection of risk drivers for human health. The complete HHRA is included as Appendix B of this RI.

6.1 EXPOSURE ASSESSMENT

This section summarizes the conceptual site model, selection of exposure pathways, screening and evaluation of COPCs, and selection of exposure parameters used in the baseline HHRA.

6.1.1 Conceptual site model and selection of exposure pathways

The conceptual site model developed in the baseline HHRA describes scenarios in which people could be exposed to COPCs associated with sediment in the LDW (Figure 6-1). The primary exposure scenarios were identified through input from site users, including local residents and members of the Muckleshoot and Suquamish Tribes, and through the review of prior risk assessments of the LDW. Exposure pathways consisted of direct contact with sediments during commercial netfishing, beach play, and clam harvesting in the LDW, as well as indirect exposure through consumption of seafood from the LDW. Exposures associated with swimming in the LDW were evaluated through the inclusion of risk estimates developed previously as part of the King County water quality assessment HHRA (1999c).



Exposure Scenarios	dermal sediment contact	incidental sediment ingestion	dermal water contact	incidental water ingestion	drinking water ingestion	fish and shellfish ingestion
Water recreation	□	□	■	■	☒	☒
Beach play	■	■	□	□	☒	☒
Occupational exposure	■	■	□	□	☒	☒
Fishing/shellfishing for consumption	■	■	□	□	☒	■

■ complete and quantified
□ complete; not quantified because of low exposure potential
☒ incomplete

Figure 6-1. Conceptual site model for baseline human health risk assessment

6.1.2 Screening and evaluation of chemicals of potential concern

This section provides a review of the COPC screening process, including the data evaluation.

6.1.2.1 Data evaluation

Sediment chemistry data used in the baseline HHRA consisted of chemical concentrations in the uppermost 15 cm of the sediment (i.e., surface sediment). The dataset included approximately 600 surface sediment samples from intertidal locations and approximately 750 surface sediment samples from subtidal locations in the LDW. The dataset consisted of data collected prior to early removal actions in the LDW in order to reflect baseline conditions. In addition, some chemistry data for tissue and sediment samples from outside the LDW were identified from other datasets or collected as part of the RI for comparison to LDW data.

Tissue data were available for English sole, starry flounder, crabs, clams, mussels, and perch caught within the LDW. Seafood consumption rates applied to these tissue data were based largely on consumption studies representative of seafood harvest from other areas of Puget Sound. Many of the species that may be consumed from these other areas (e.g., speckled sanddab, Pacific cod, rockfish, spiny dogfish, walleye pollock) are rarely found in the LDW, and no LDW tissue chemistry data were available for those species. Therefore, data from a representative trophic group were used as a surrogate. In addition, although salmon are a highly preferred and consumed fish from the LDW and tissue data were available for salmon, human health risks were not calculated for the consumption of adult salmon. The exposure of juvenile salmon to risk driver chemicals in LDW sediment is not anticipated to significantly influence chemical concentrations in tissues of adult salmon. PCBs in juvenile salmon accumulated during the time they spend in the LDW constitute a small fraction of the PCBs in adult salmon. Most of the PCBs in adult salmon result from foraging activities in Puget Sound or the Pacific Ocean. Consequently, site-related PCB human health risks associated with salmon consumption were not estimated. Although risks from salmon consumption were not included in the baseline HHRA, risks to juvenile salmon were addressed in the ERA (Appendix A).

Sediment and tissues samples were analyzed for numerous chemicals, including metals, SVOCs, organochlorine pesticides, and PCBs. Most organochlorine pesticides in seafood tissues collected in 2004 were qualified as estimated concentrations with uncertain presence. The tissue concentrations used to determine the intake of these chemicals were thus highly uncertain. There is uncertainty as to whether these chemicals were present; however, if present, their concentrations were likely biased high because of PCB interferences. Therefore, organochlorine pesticide data may not accurately represent exposures to these chemicals from seafood consumption within the LDW. Uncertainties in the data used in this assessment are summarized in Section 4.2.10.3 and in Appendix B, Section B.6.1.

6.1.2.2 COPC screening

Using the dataset defined for the baseline HHRA, a risk-based screen was performed following EPA guidance to identify COPCs to be evaluated. Briefly, the maximum concentrations in sediment were screened against risk-based concentrations (RBCs) for soil provided by EPA Region 9 (EPA 2004). The maximum seafood tissue concentrations were screened against EPA Region 3 RBCs (EPA 2005c) for fish tissue, which had been adjusted to account for tribal consumption rates (97.5 g/day) (EPA 2006c) and a lower target HQ of 0.1, as required by EPA Region 10 (EPA 1996). Sixty-four chemicals were identified as COPCs in sediment or tissue for one or more exposure scenarios. Many of these 64 chemicals were identified for more detailed analysis using both seafood consumption and direct sediment exposure scenarios (Table 6-1). Of the 64 COPCs, 6 chemicals were never detected in tissue (but were detected in sediment), and 20 were never detected in either sediment or tissue.[116] These 26 chemicals were included as COPCs because analytical RLs were above the screening criteria for tissue, sediment, or both. The undetected COPCs were evaluated in the uncertainty analysis: 26 of these COPCs were evaluated for seafood consumptions scenarios, and 5 were evaluated for direct-contact scenarios.

Table 6-1. COPCs identified for direct-contact sediment exposure and seafood consumption scenarios

CHEMICAL	NETFISHING SCENARIOS	BEACH PLAY AND CLAMMING SCENARIOS	SEAFOOD CONSUMPTION SCENARIOS
Chemicals detected in both tissue and sediment			
Inorganic			
Aluminum[a]	X	X	
Antimony	X	X	X
Arsenic	X	X	X[b]
Barium[a]	X	X	
Cadmium	X	X	X
Chromium	X	X	X
Copper	X	X	X
Iron[a]	X	X	
Lead	X	X	X[c]
Manganese[a]	X	X	
Mercury		X	X
Molybdenum		X	
Nickel			X
Silver		X	
TBT			X
Thallium	X	X	
Vanadium	X	X	X
Zinc		X	X
Organic			
4-methylphenol			X
Aldrin			X
alpha-BHC			X

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[116] N-nitroso-di-n-propylamine was not included in the risk characterization but was discussed in the uncertainty analysis despite being detected in one tissue sample. This chemical was never detected in sediment samples, and had high RLs in tissue samples. The detection in tissue was qualified as JN (estimated concentration, tentative identification).

CHEMICAL	NETFISHING SCENARIOS	BEACH PLAY AND CLAMMING SCENARIOS	SEAFOOD CONSUMPTION SCENARIOS
beta-BHC			X
gamma-BHC			X
BEHP			X
BBP			X
Carbazole			X
cPAHs	X	X	X
Total chlordane			X
Total DDTs		X	X
Dieldrin	X	X	X
Dioxin and furan TEQ[a]	X	X	X[d]
Endrin			X
Endrin aldehyde			X
Heptachlor			X
Heptachlor epoxide			X
Hexachlorobenzene			X
PCB TEQ	X	X	X
Total PCBs	X	X	X
Pentachlorophenol			X
Chemicals detected in sediment, but not detected in tissue			
1,3-Dichlorobenzene			X_{ND}
1,4-Dichlorobenzene			X_{ND}
Aniline[e]			X_{ND}
Hexachlorocyclopentadiene			X_{ND}[e]
n-Nitrosodiphenylamine			X_{ND}
Toxaphene	X	X	X_{ND}
Chemicals not detected in sediment or tissue			
1,2-Diphenylhydrazine			X_{ND}
2,4,6-Trichlorophenol			X_{ND}
2,4-Dichlorophenol			X_{ND}
2,4-Dinitrophenol			X_{ND}
2,4-Dinitrotoluene			X_{ND}
2,6-Dinitrotoluene			X_{ND}
2-Chlorophenol			X_{ND}
3,3'-Dichlorobenzidine			X_{ND}
3-Nitroaniline			X_{ND}
4,6-Dinitro-o-cresol		X_{ND}	X_{ND}

CHEMICAL	NETFISHING SCENARIOS	BEACH PLAY AND CLAMMING SCENARIOS	SEAFOOD CONSUMPTION SCENARIOS
4-Chloroaniline			X_{ND}
4-Nitroaniline			X_{ND}
Benzidine	X_{ND}	X_{ND}	X_{ND}
Bis(2-chloroethyl)ether		X_{ND}	X_{ND}
Bis(2-chloroisopropyl)ether			X_{ND}
Hexachlorobutadiene			X_{ND}
Hexachloroethane			X_{ND}
Nitrobenzene			X_{ND}
n-Nitrosodimethylamine	X_{ND}	X_{ND}	X_{ND}
n-Nitroso-di-n-propylamine		X_{ND}	X_{ND}[f]

[a] This chemical was not analyzed in tissue.

[b] For the seafood consumption scenarios, the arsenic assessment was based on inorganic arsenic exposure and toxicity data.

[c] No RBC was available for lead. An alternative toxicity evaluation method recommended by EPA was used instead (see Appendix B, Section B.3.4.4, for more information on lead models).

[d] Dioxin and furan TEQ was identified as a COPC in sediment and tissue; although dioxins and furans were not analyzed in tissue, they were identified as potential bioaccumulative compounds by EPA (EPA 2000a).

[e] This chemical was detected in one sediment sample.

[f] One composite sample of whole-body English sole contained a detected concentration of n-nitroso-di-n-propylamine (270 µg/kg ww). However, this result was qualified as JN (estimated concentration, tentative identification). Given the uncertain quantification for this single result (all other results were undetected), the risks for this chemical were discussed in the uncertainty analysis.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
BHC – benzene hexachloride
COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

X indicates that the chemical was selected as a COPC and was detected in the exposure scenario media.
X_{ND} indicates that the chemical was selected as a COPC but was not detected in the exposure scenario media.

6.1.3 Exposure scenarios and associated exposure parameters

Human exposures to COPCs were estimated for several seafood consumption and direct-contact sediment exposure scenarios, as described below. Exposure estimates were based on the concentration data for each COPC and health-protective assumptions regarding the rates and frequency of seafood intake and the rates, frequency, and duration of direct sediment contact associated with incidental ingestion and dermal exposure.

The exposure scenarios that were developed are based on assumptions that cover a range of expected behavior. At the high end, EPA (1989) defines the reasonable maximum exposure (RME) as “the highest exposure that is reasonably expected to occur at a site.” The scenarios developed in the baseline HHRA are consistent with RME guidelines. Average exposures, on the other hand, are referred to as central tendency (CT) scenarios.

Another method of examining exposure is to identify a unit of exposure that a member of the public can use to assess the risks associated with their individual behavior. This last approach was used to characterize seafood consumption exposure on an individual basis. The unit of exposure used in this risk assessment was one meal per month. The one-meal-per-month exposure scenario is not meant to actually describe behavior that is occurring on the LDW because there are no actual data on current seafood consumption rates for the LDW.

EPA generally uses RME scenarios to evaluate the need for remedial actions at a site (EPA 1989). RME, by definition, likely overestimates exposure for many individuals. CT exposures are useful for characterizing uncertainty in exposure and risks (National Research Council 1994). However, average exposure estimates are not favored in decision-making because they will underestimate exposure for a substantial number of individuals (EPA 1989).

6.1.3.1 Seafood consumption scenarios

Several different seafood consumption scenarios were evaluated, as summarized in Table 6-2. Seafood consumption rates that were assumed for the seafood consumption scenarios with multiple species were developed by EPA based on data collected from several surveys. Specifically, for the adult tribal RME seafood consumption scenario based on Tulalip data, a consumption rate of 194 g per day was considered However, given that adult salmon were deemed not to have a significant site-related contribution of PCBs and other risk driver chemicals (i.e., most of their chemical accumulation results from exposures outside of the LDW), only the consumption of resident seafood was used for the risk assessment. Therefore, a resident seafood consumption rate of 97.5 g per day (which equals approximately three 227-g [8-oz] meals per week)[117] was assumed, based on EPA's analysis (*Framework for Selecting and Using Tribal Fish and Shellfish Consumption Rates for Risk-Based Decision Making at CERCLA and RCRA Cleanup Sites in Puget Sound and the Strait of Georgia* [EPA 2007]) of a survey of the Tulalip Tribes' consumption of resident species from the Puget Sound region (Toy et al. 1996). This rate was intended to reflect the 95th percentile of Puget Sound non-anadromous seafood consumption and was designated as a RME scenario. For some pathways, both RME exposure estimates and CT exposure estimates were developed to describe the range of possible exposures and risks, as discussed above. Specific assumptions for the CT scenarios were developed in consultation with EPA (EPA 2007d; Kissinger 2005).

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[117] Rate does not include consumption of anadromous fish. Total consumption rate including anadromous fish is 194 g/day (EPA 2007d).

Table 6-2. Summary of seafood consumption scenarios

SCENARIO	INGESTION RATE (g/day)					EXPOSURE DURATION (years)
	PELAGIC FISH	BENTHIC FISH	CRABS	OTHER SHELLFISH	TOTAL[a]	
Adult tribal RME (Tulalip data)	8.1	7.5	43.4[b]	38.5[b]	97.5	70
Adult tribal CT (Tulalip data)	1.3	1.2	6.6[b]	5.9[b]	15.0	30
Child tribal RME (Tulalip data)	3.2	3.0	17.4[b]	15.4[b]	39.0	6
Child tribal CT (Tulalip data)	0.52	0.48	2.64[b]	2.34[b]	6.0[c]	6
Adult tribal (Suquamish data)	56	29.2[c]	55[c]	444[c]	584.2[c]	70
Adult API – RME	4.9	2.4	10.6	33.6	51.5	30
Adult API – CT	0.5	0.24	1.1	3.5	5.3	12
Adult one meal per month[d]	7.5	7.5	7.5	7.5	na	30

[a] Rates do not include consumption of anadromous fish.

[b] This table presents the same consumption rates based on the Tulalip Tribes survey as were previously provided by EPA and used in the LDW HHRA, Appendix B. These rates were also used in the risk results summarized in this section. The apportionment of shellfish (i.e., the amount of crab consumed relative to other shellfish, but not the total quantity consumed) for scenarios based on the Tulalip Tribes survey has since been updated per a correction provided by EPA (2009). The influence of this correction on the total risk estimates is relatively minor. This change and its impact on risk estimates are described in detail in an errata to Appendix B. The updated consumption rates are provided in Section 8 and were used in the development of RBTCs.

[c] Totals of benthic and shellfish categories (and overall total) differ slightly from values provided by EPA (2007d) because of significant figure and rounding issues when total consumption was allocated to more specific consumption categories. Overall total listed here is slightly higher than the corresponding value provided by EPA (2007d).

[d] Adult one-meal-per-month consumption was evaluated for individual seafood categories independently to reflect different fishing and consumption practices.

API – Asian and Pacific Islander
CT – central tendency
na – not applicable
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure

The consumption rate for the adult tribal RME scenario based on Tulalip data was assumed to represent the consumption of seafood caught from the LDW and was further divided into seafood categories as shown in Table 6-2. It is likely that current seafood consumption rates within the LDW are lower than those documented in the Tulalip tribal study because of existing seafood consumption advisories or general concerns about contamination.

EPA's Superfund risk assessment guidance (EPA 1989) requires that exposure estimates be protective of future uses. Tribes with treaty rights to obtain seafood from the LDW may increase their seafood consumption rate in the future as chemical contamination in the LDW decreases. Habitat improvements may also increase the harvestable population of fish and shellfish to some degree. Consequently, the RME seafood consumption rates evaluated in the baseline HHRA were intended to be protective of both current and future uses.

Health risks were also quantified for several seafood consumption scenarios[118] in addition to the adult tribal RME scenario based on Tulalip data:

- Two Asian and Pacific Islander (API) scenarios (5.3 and 51.5 g/day of resident seafood for CT and RME, respectively (Kissinger 2005)
- An adult tribal CT scenario based on Tulalip data (15 g/day of resident seafood) (Hiltner 2007)
- Two child tribal scenarios based on Tulalip data (8.6 and 55.9 g/day of resident seafood for CT and RME, respectively) (EPA 2006d, 2007c; Hiltner 2007)
- A second adult tribal scenario based on Suquamish data (584.2 g/day of resident seafood) (EPA 2007d)
- Four adult one-meal-per-month consumption scenarios based on individual seafood categories (7.5 g/day)

Table 6-2 presents the ingestion rates and exposure durations associated with the various seafood consumption categories.

"Based on policy considerations, EPA is intending to use the Tulalip Tribes' seafood consumption rate as the principal rate to compute health protective tribal seafood consumption risks" for the LDW (EPA 2006d). The Suquamish seafood consumption scenario was included at the request of the Suquamish and Muckleshoot Indian Tribes to assist in characterizing the potential range of seafood consumption risks, although EPA (2005a) has questioned whether the high Suquamish rate could be sustained within the LDW.

The adult tribal seafood consumption scenarios based on Tulalip data, child tribal seafood consumption scenarios based on Tulalip data, adult tribal seafood consumption scenario based on Suquamish data, and adult API seafood consumption scenarios included a combination of all the seafood categories. For the adult one-meal-per-month scenario, risk was evaluated based on a consumption rate of one meal per month of pelagic fish (such as perch), benthic fish fillets (such as English sole and starry flounder), crab edible meat, or clams. The one-meal-per-month scenario was included for risk communication to enable individuals to estimate the risk associated with specific behaviors, rather than to describe the behavior of any particular group. Consistent with EPA risk assessment guidance, all assumptions regarding the amount of seafood ingested were selected to be health-protective to avoid underestimating risks. Consequently, individual risk estimates are more likely to be overestimates for most chemicals.

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[118] Rates do not include consumption of anadromous fish. Total consumption rates for tribal and API scenarios were higher than values presented here when anadromous fish were included (EPA 2007d; Kissinger 2005).

6.1.3.2 Direct sediment exposure scenarios

Risks associated with sediment skin contact or incidental sediment ingestion were quantified in the baseline HHRA for adults engaging in clamming or tribal netfishing and for children playing in intertidal sediment areas (hereafter referred to as "beach play"). Exposure frequency and duration assumptions for the evaluation of direct sediment exposure under the commercial netfishing scenario were based on site use information collected from the Muckleshoot Indian Tribe, which conducts commercial netfishing for adult salmon within the LDW. Exposure parameter values for the beach play and clam harvesting scenarios were based primarily on EPA guidance, a King County lakes recreational survey (applied to beach play scenario only) (Parametrix 2003), a recent EPA memo on tribal clamming (Kissinger 2007), and best professional judgment because site-specific data on exposure frequency and duration for these scenarios were not available. Table 6-3 provides a summary of the exposure parameters for these scenarios.

Table 6-3. Summary of sediment exposure scenarios

SCENARIO	INCIDENTAL SEDIMENT IR (g/day)	EXPOSURE FREQUENCY (days/year)	EXPOSURE DURATION (years)	SKIN SURFACE AREA EXPOSED (cm^2)
Netfishing RME	0.05	119	44	3,600
Netfishing CT	0.05	63	29	3,600
Child beach play RME[a]	0.2	65	6	varies with age
Clamming (7 days per year)	0.1	7	30	6,040
Tribal clamming RME (120 days per year)	0.1	120	64	6,040
Tribal clamming (183 days per year)	0.1	183	70	6,040

[a] For the beach play scenarios, the LDW was divided into eight areas to assess risks associated with different parts of the LDW. Children were evaluated from birth through 6 years of age.

CT – central tendency

IR – ingestion rate

RME – reasonable maximum exposure

6.1.3.3 Exposure point concentrations

Exposure point concentrations (EPCs) for COPCs are intended to represent health-protective estimates of long-term exposure concentrations for a given scenario. In the baseline HHRA, the EPC was one of the following:

- If there were at least six detected concentrations or calculated totals, the EPC was the UCL[119] (calculated using EPA ProUCL software, Version 4).

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[119] The 95% UCL was typically used; but in some cases, 97.5% or 99% UCLs were recommended by ProUCL and were used for risk calculations.

- If there were one to five detected concentrations, the EPC was the higher of one-half the maximum RL or the maximum detected value. For calculated totals, such as total PCBs, cPAHs, and dioxins/furans, the EPC was the higher of the maximum detected value or one-half the maximum non-detect value reported. The methods used for calculating these totals are presented in Section 4.1.4.
- If the chemical was not detected in any samples, one-half the maximum RL (or the maximum non-detect value for calculated totals) was selected as the EPC.

All concentrations qualified as estimates (i.e., J-qualified data) were assumed to indicate positive identification of the chemical and were used without modification in subsequent calculations. Some J-qualified data, most notably the 2004 pesticide results, were also N-qualified, indicating a tentative identification of the chemical. In the case of pesticides, the N-qualifier was necessary because of the high potential for analytical interference from PCBs (Windward 2005b). JN-qualified data were used in the risk assessment, but the uncertainty associated with these results was higher than the uncertainty associated with J-qualified results. Accordingly, in the risk characterization (Appendix B, Section B.5), the JN-qualified organochlorine pesticide data were presented separately from data for chemicals that were not JN-qualified. Analytical results for organochlorine pesticides used in the baseline HHRA from events prior to 2004 were likely to also have been complicated by interference from PCBs; however, the detected results were not all JN-qualified.

The EPCs for the direct sediment exposure scenarios (i.e., netfishing, beach play, and clam harvesting) were calculated for the sediment area over which exposure could potentially occur. The netfishing scenario assumed that tribal members who engage in commercial netfishing could be exposed to both intertidal and subtidal sediment adhering to their nets; consequently, the entire baseline surface sediment dataset was used for EPC calculation for this scenario.

For the beach play scenario, EPCs were based on data for intertidal sediment (i.e., sediment periodically exposed to air during low tides) from areas that are accessible to the public. The LDW was divided into eight areas for the beach play scenarios, corresponding to eight contiguous areas where access to intertidal sediments is relatively easy. Separate risk estimates were made for each of those eight areas. The areal extent of each beach, EPCs, and risks associated with the beach play scenario are presented in Map 6-1. The exposure frequency selected for the beach play scenario (i.e., 65 days per year) was based on King County's survey of parks immediately adjacent to lakes (Parametrix 2003) and represents the 95th percentile of exposure frequency for children up to 6 years old who play in sand near the water. This behavior is consistent with the behavior that is assumed for the beach play scenario in the LDW.

The clam harvesting scenarios assumed that people are exposed to COPCs in sediment as they dig for clams. The two tribal clamming scenarios (RME and 183 days per year) included all potential clam habitat areas (as identified during a 2004 survey) that could be accessed either by boat or on foot from the bank. Another clamming scenario used a lower exposure frequency (7 days per year) and included only potential clam habitat areas that could be accessed from the bank (Map 6-2).

EPCs for the seafood consumption scenarios were calculated separately for various types of seafood, or consumption categories. Seven consumption categories were developed based on available seafood tissue types: pelagic fish, benthic fish—fillet, benthic fish—whole body, crab—edible meat, crab—whole body, clams, and mussels. In some cases, chemistry data for more than one species were combined within a single consumption category (e.g., data for Dungeness crab and slender crab edible meat were combined in the edible crab meat category). A COPC intake rate was then calculated for each consumption category using the COPC tissue dataset and the consumption rate for each category. The chemical intakes for each consumption category were then summed within each seafood consumption scenario (except the adult one-meal-per-month scenario) to yield an overall COPC intake for that scenario.

6.2 TOXICITY ASSESSMENT

EPA toxicity values (i.e., slope factors [SFs] for the evaluation of carcinogenic risks or reference doses [RfDs] for the evaluation of effects other than cancer) were identified for all COPCs. Toxicity values for each COPC have been established by EPA and other agencies and are based on either laboratory experiments using animals or epidemiological studies of human populations unintentionally exposed in the workplace or the environment. The SFs provide a health-protective means to evaluate risks because they represent upper bound estimates of carcinogenic potency. Similarly, non-cancer toxicity values (i.e., RfDs) are health-protective in that they are typically derived based on the most sensitive endpoint and population for which adequate data are available and include uncertainty factors or extrapolations to account for sensitive sub-populations or other limitations of the toxicity study data on which they were based. A detailed discussion of the toxicity values used in the HHRA is presented in Appendix B, Section B.4.

6.3 RISK CHARACTERIZATION

This section presents the risk estimates calculated using the exposure parameters and toxicity values discussed in Sections 6.1 and 6.2, respectively.

6.3.1 Risk estimate calculations

Carcinogenic risks and non-carcinogenic health effects were evaluated separately in the baseline HHRA because of fundamental differences in assumptions about the mechanism of these toxic effects. Carcinogenic risk estimates were calculated by multiplying the estimated chemical intake by the SF. Excess cancer risk estimates were compared with EPA's acceptable risk range of 10^{-6} to 10^{-4} established in the National Contingency Plan for Superfund sites (40 CFR 300). The lifetime risk of developing cancer in the US population is one in two (i.e., 5×10^{-1}) for men and one in three (i.e., 3×10^{-1}) for women (American Cancer Society 2006). A 1×10^{-6} excess cancer risk represents an additional one-in-one-million probability that an individual may develop cancer over a 70-year lifetime as a result of exposure to chemicals in LDW sediments and surface water (either through direct exposure or indirect exposure through the consumption of seafood).

Carcinogenic risks associated with PCBs may be assessed based on either the dose of total PCBs and a PCB slope factor or the dose of a subset of PCB congeners that have a chemical structure and toxicity similar to that of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) and the TCDD slope factor**.** The toxicity of these dioxin-like PCB congeners is evaluated in terms of PCB TEQs. The TEQ relates the toxicity of coplanar PCBs to the toxicity TCDD. Risk estimates from the two approaches may be partially overlapping in terms of both exposure and toxicity. There is currently no consensus approach for considering these two risk approaches together. Some risk assessments have added these risks together to evaluate overall PCB risks, likely overestimating PCB risks. Other risk assessments have set overall PCB risks at either the risk calculated using the overall PCB dose and PCB Aroclor slope factor or the risk calculated using the PCB TEQ dose and dioxin TCDD slope factor. This latter approach may underestimate PCB risks. In this risk assessment, two different total excess cancer risk estimates were provided: one estimate included all COPC risks except those for PCB TEQ (i.e., including risks from all non-PCB COPCs plus total PCBs); the other estimate included all COPCs except total PCBs (i.e., including risks from all non-PCB COPCs plus PCB TEQ).

Chemicals with non-carcinogenic health effects are generally not toxic below a certain threshold; a critical chemical dose must be exceeded before adverse health effects are observed. The potential for non-carcinogenic health effects is represented by the ratio of the estimated chemical intake to the critical chemical dose (called a reference dose) and is expressed as an HQ. Exposures resulting in an HQ less than or equal to 1 are unlikely to result in non-cancer adverse health effects. For chemicals sharing a common toxicity endpoint, the HQs were summed to develop a hazard index (HI) for that endpoint.

6.3.2 Risk characterization results

Tables 6-4 to 6-6 summarize the excess cancer risk estimates and non-cancer hazards for chemicals that exceeded 1×10^{-6} excess cancer risk or an HQ of 1 for at least one scenario. All other risk estimates are presented in Appendix B, Section B.5.

Estimated excess cancer risk estimates were highest for the seafood consumption scenarios (Table 6-4). The cumulative risk for all carcinogenic chemicals was 3×10^{-3} for the adult tribal RME seafood consumption scenario based on Tulalip data; the primary contributors were PCBs (2×10^{-3}) and inorganic arsenic (1×10^{-3}). The cumulative excess cancer risk for the seafood consumption scenarios did not include dioxins and furans, which were not analyzed in tissue samples. It was assumed that if dioxins and furans had been analyzed and detected in tissue samples, risks would have been unacceptable. The majority of the excess cancer risk for PCBs was attributable to consumption of pelagic fish and clams. The excess cancer risk estimates from inorganic arsenic were largely attributable to consumption of clams. Inorganic arsenic concentrations in fish and crabs were much lower and were very similar to background tissue concentrations.

Cumulative excess cancer risk estimates for the other seafood consumption scenarios (except for the adult tribal scenario based on Suquamish data) were lower (approximately 1.3 to 33% of cancer risk estimates for the adult tribal RME scenario based on Tulalip data) (Table 6-4). Specifically, the total excess cancer risks were 2 × 10^{-5} to 1 × 10^{-3} for the API consumption scenarios; 1 × 10^{-4} for the adult tribal CT scenario based on Tulalip data; 6 × 10^{-5} to 8 × 10^{-4} for the child tribal scenarios based on Tulalip data; 3 × 10^{-2} for the adult tribal scenario based on Suquamish data; and 2 × 10^{-4} (pelagic fish), 1 × 10^{-4} (benthic fish fillets), 4 × 10^{-5} (crab edible meat), and 2 × 10^{-4} (clams) for individuals who consumed those seafood categories at a rate of one meal per month. The risks for the adult tribal scenario based on Suquamish data were 10 times higher than risks for the adult tribal RME scenario based on Tulalip data, reflecting the much higher seafood consumption rate (almost three 8-oz meals per day) used in the adult Suquamish scenario estimates. In the evaluation of non-cancer hazards, arsenic and PCBs had HQs greater than 1 for one or more of the RME seafood consumption scenarios, indicating some potential for adverse effects other than cancer. In addition, HQs were greater than 1 for the child tribal RME scenario based on Tulalip data for TBT and vanadium (Table 6-5). For the adult tribal scenario based on Suquamish data, two additional chemicals had HQs greater than 1: chromium and mercury. HQs for these chemicals were less than 1 for all other seafood consumption scenarios. For the purpose of brevity, chemical-specific HQ estimates are provided in Table 6-5 only for chemicals exceeding an HQ of 1 for any scenario.

Excess cancer risk estimates for the direct sediment exposure scenarios were much lower than those for the seafood consumption scenarios. With the exception of the tribal clamming RME scenario and the tribal clamming 183-day-per-year scenario, all excess cancer risk estimates for direct sediment exposure scenarios were less than or equal to 5 × 10^{-5} (i.e., only 1.7% of the risks from the adult tribal RME seafood consumption scenario based on Tulalip intake data) (Table 6-6). Total excess cancer risk from the tribal clamming RME scenario was 1 × 10^{-4} (excluding PCB TEQ), which is 3.3% of the risks from the adult tribal RME seafood consumption scenario based on Tulalip data and 2 × 10^{-4} (excluding PCB TEQ) for the tribal clamming 183-day-per-year scenario. Total excess cancer risk estimates were greater than 1 × 10^{-6} for the netfishing scenarios and between 5 × 10^{-6} and 5 × 10^{-5} for the beach play RME scenarios. Risk estimates associated with specific beach play areas are also presented in Map 6-1. The chemicals contributing the most to direct sediment excess cancer risk estimates were dioxins and furans, PCBs, arsenic, and cPAHs. No HQs were greater than 1 for any of the direct sediment exposure scenarios.

Table 6-4. Summary of estimated excess cancer risks for the seafood consumption scenarios

Chemical	Adult Tribal RME (Tulalip Data)	Adult Tribal CT (Tulalip Data)	Child Tribal RME (Tulalip Data)	Child Tribal CT (Tulalip Data)	Adult Tribal (Suquamish Data)	Adult API RME	Adult API CT	Adult One Meal per Month			
								Benthic Fish	Clam	Crab	Pelagic Fish
Arsenic (inorganic) [a]	1×10^{-3}	6×10^{-5}	3×10^{-4}	3×10^{-5}	2×10^{-2} [b]	7×10^{-4}	1×10^{-5}	4×10^{-7}	1×10^{-4}	3×10^{-6}	6×10^{-6}
BEHP	6×10^{-6}	2×10^{-7}	1×10^{-6}	7×10^{-8}	4×10^{-5}	2×10^{-6}	3×10^{-8}	8×10^{-7}	8×10^{-8}	8×10^{-8} [c]	1×10^{-6}
cPAHs[d]	7×10^{-5}	4×10^{-6}	7×10^{-5}	8×10^{-6}	8×10^{-4}	3×10^{-5}	8×10^{-7}	2×10^{-7}	7×10^{-6}	2×10^{-7}	3×10^{-7}
Dioxin and furan TEQ[e]	na	na	na	na	na	na	na	na	na	na	na
PCB TEQ	1×10^{-3}	6×10^{-5}	2×10^{-4}	2×10^{-5}	7×10^{-3}	4×10^{-4}	7×10^{-6}	8×10^{-5}	2×10^{-5}	2×10^{-5}	2×10^{-4}
Total PCBs	2×10^{-3}	6×10^{-5}	3×10^{-4}	3×10^{-5}	1×10^{-2} [b]	5×10^{-4}	8×10^{-6}	1×10^{-4}	5×10^{-5}	2×10^{-5}	2×10^{-4}
Pentachlorophenol [a]	9×10^{-5} [f]	2×10^{-6} [f]	2×10^{-5} [f]	7×10^{-7} [f]	5×10^{-4} [f]	2×10^{-5}	3×10^{-7}	2×10^{-5} [c]	1×10^{-6} [c]	2×10^{-6} [c]	1×10^{-5}
Subtotal (excluding PCB TEQ)	**3×10^{-3}**	**1×10^{-4}**	**7×10^{-4}**	**7×10^{-5}**	**3×10^{-2}**	**1×10^{-3}**	**2×10^{-5}**	**1×10^{-4}**	**2×10^{-4}**	**3×10^{-5}**	**2×10^{-4}**
Subtotal (excluding total PCBs)	**2×10^{-3}**	**1×10^{-4}**	**6×10^{-4}**	**6×10^{-5}**	**3×10^{-2}**	**1×10^{-3}**	**2×10^{-5}**	**1×10^{-4}**	**1×10^{-4}**	**3×10^{-5}**	**2×10^{-4}**
Tentatively Identified Chemicals (JN-qualified)											
Aldrin	5×10^{-5} [f]	1×10^{-6} [f]	9×10^{-6} [f]	6×10^{-7} [f]	2×10^{-4}	1×10^{-5}	2×10^{-7}	3×10^{-6} [c]	8×10^{-7} [c]	3×10^{-6} [c]	3×10^{-6}
alpha-BHC	2×10^{-5} [f]	5×10^{-7} [f]	3×10^{-6} [f]	2×10^{-7} [f]	6×10^{-5}	3×10^{-6}	6×10^{-8}	1×10^{-6}	1×10^{-7}	1×10^{-6} [c]	1×10^{-6}
beta-BHC	6×10^{-6} [f]	3×10^{-7} [f]	1×10^{-6} [f]	1×10^{-7} [f]	3×10^{-5}	1×10^{-6}	3×10^{-8}	3×10^{-7}	1×10^{-7}	3×10^{-7} [c]	6×10^{-7}
Carbazole	5×10^{-5}	1×10^{-6}	8×10^{-6}	4×10^{-7}	2×10^{-4}	1×10^{-5}	8×10^{-8}	1×10^{-6} [c]	9×10^{-8} [c]	1×10^{-6} [c]	1×10^{-5}
Total chlordane	6×10^{-6}	2×10^{-7}	1×10^{-6}	9×10^{-8}	3×10^{-5}	2×10^{-6}	3×10^{-8}	3×10^{-7}	7×10^{-8}	7×10^{-8}	1×10^{-6}
Total DDTs	2×10^{-5}	1×10^{-6}	4×10^{-6}	4×10^{-7}	1×10^{-4}	6×10^{-6}	1×10^{-7}	1×10^{-6}	2×10^{-7}	4×10^{-7}	4×10^{-6}
Dieldrin	1×10^{-4}	3×10^{-6}	2×10^{-5}	1×10^{-6}	1×10^{-3}	5×10^{-5}	4×10^{-7}	3×10^{-6} [c]	9×10^{-6}	3×10^{-6}	3×10^{-6} [c]
gamma-BHC	6×10^{-6}	1×10^{-7}	1×10^{-6}	5×10^{-8}	3×10^{-5}	1×10^{-6}	1×10^{-8}	2×10^{-7} [c]	1×10^{-7}	2×10^{-7}	1×10^{-7}
Heptachlor	1×10^{-5} [f]	4×10^{-7} [f]	3×10^{-6} [f]	2×10^{-7} [f]	6×10^{-5}	3×10^{-6}	4×10^{-8}	7×10^{-7} [c]	1×10^{-7} [c]	7×10^{-7} [c]	2×10^{-6}
Heptachlor epoxide	3×10^{-5}	1×10^{-6}	6×10^{-6}	5×10^{-7}	2×10^{-4}	9×10^{-6}	1×10^{-7}	1×10^{-6} [c]	6×10^{-7}	9×10^{-7}	4×10^{-6}
Hexachlorobenzene	1×10^{-5}	2×10^{-7}	2×10^{-6}	1×10^{-7}	4×10^{-5}	2×10^{-6}	3×10^{-8}	6×10^{-7}	6×10^{-8}	6×10^{-7}	9×10^{-7}
Subtotal	**3×10^{-4}**	**9×10^{-6}**	**6×10^{-5}**	**4×10^{-6}**	**2×10^{-3}**	**1×10^{-4}**	**1×10^{-6}**	**1×10^{-5}**	**1×10^{-5}**	**1×10^{-5}**	**3×10^{-5}**
Total excess cancer risk (excluding PCB TEQ)	**3×10^{-3}**	**1×10^{-4}**	**8×10^{-4}**	**7×10^{-5}**	**3×10^{-2}**	**1×10^{-3}**	**2×10^{-5}**	**1×10^{-4}**	**2×10^{-4}**	**4×10^{-5}**	**2×10^{-4}**
Total excess cancer risk (excluding total PCBs)	**2×10^{-3}**	**1×10^{-4}**	**7×10^{-4}**	**6×10^{-5}**	**3×10^{-2}**	**1×10^{-3}**	**2×10^{-5}**	**1×10^{-4}**	**1×10^{-4}**	**4×10^{-5}**	**2×10^{-4}**

[a] No mussel data were available for this chemical. When calculating the chronic daily intake and risk values, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

[b] Because the excess cancer risk is greater than or equal to 0.01, risk was calculated using the exponential equation in EPA (1989).

[c] No detected values in this seafood category. Chronic daily intake and risk estimate are based on one-half the maximum reporting limit.

[d] cPAH concentrations are based on benzo(a)pyrene equivalents. Data used in the risk characterization are from only 2004 because of high reporting limits in historical data. All cPAH data are analyzed in the uncertainty analysis (Appendix B, Section B.6). Because of the potential for increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (EPA 2005e), the risk estimate for children for cPAHs is based on dose adjustments across the 0-to-6-year age range of children. See Appendix B, Section B.5.1, for more information.

[e] Tissue data for dioxins and furans were not collected. Thus, the calculated risk total, which does not include risks from dioxins and furans, is underestimated to an unknown degree.

[f] Greater than 50% of the risk associated with this chemical is derived from seafood categories with no detected values.

API – Asian and Pacific Islander
BEHP – bis(2-ethylhexyl) phthalate
BHC – benzene hexachloride
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
na – not available
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TEQ – toxic equivalent

Table 6-5. Summary of estimated non-cancer hazards for the seafood consumption scenarios

Chemical	Adult Tulalip RME (Tulalip Data)	Adult Tribal CT (Tulalip Data)	Child Tribal RME (Tulalip Data)	Child Tribal CT (Tulalip Data)	Adult Tribal (Suquamish Data)	Adult API RME	Adult API CT	Adult One Meal per Month			
								Benthic Fish	Clam	Crab	Pelagic Fish
Arsenic[a, b]	3	0.3	7	0.7	38	3	0.2	0.002	0.7	0.01	0.03
Chromium	0.2	0.02	0.3	0.04	2	0.1	0.01	0.002	0.03	0.006	0.007
Mercury	0.5	0.07	1	0.1	2	0.3	0.02	0.06	0.02	0.07	0.04
Total PCBs	40	4	86	8	274	29	2	6	3	1	10
TBT (as ion)	1	0.2	3	0.3	15	1	0.1	0.002	0.3	0.02	0.06
Vanadium	0.8	0.1	2	0.2	9	0.8	0.07	0.01	0.2	0.01	0.06
Hazard Indices by Effect[c]											
HI for cardiovascular endpoint[d]	**4**	**0.4**	**9**	**0.9**	**47**	**4**	**0.3**	**0.01**	**0.9**	**0.02**	**0.09**
HI for developmental endpoint[e]	**41**	**4**	**87**	**8**	**276**	**29**	**2**	**6**	**3**	**1**	**10**
HI for hematologic endpoint[f]	**0.2**	**0.03**	**0.5**	**0.05**	**2**	**0.2**	**0.01**	**0.006**	**0.03**	**0.01**	**0.009**
HI for immunological endpoint[g]	**41**	**4**	**89**	**8**	**289**	**30**	**2**	**6**	**3**	**1**	**10**
HI for kidney endpoint[h]	**0.4**	**0.05**	**1.0**	**0.1**	**2**	**0.3**	**0.02**	**0.03**	**0.03**	**0.04**	**0.04**
HI for liver endpoint[i]	**1**	**0.1**	**3**	**0.3**	**7**	**0.8**	**0.05**	**0.1**	**0.1**	**0.09**	**0.3**
HI for neurological endpoint[j]	**41**	**4**	**87**	**8**	**276**	**29**	**2**	**6**	**3**	**1**	**10**
HI for dermal endpoint[k]	**3**	**0.3**	**7**	**0.7**	**38**	**3**	**0.2**	**0.01**	**0.7**	**0.02**	**0.06**

[a] No mussel data were available for this chemical. When calculating the risk values, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

[b] Arsenic risk estimates are based on inorganic arsenic.

[c] HIs include risks associated with all COPCs by endpoint. However, only those COPCs with a hazard quotient greater than or equal to 1 for at least one scenario are listed in this table.

[d] Cardiovascular endpoint is for arsenic and vanadium.

[e] Developmental endpoint is for PCBs and mercury.

[f] Hematologic endpoint is for antimony and zinc.

[g] Immunological endpoint is for PCBs and TBT.

[h] Kidney endpoint is for 4-methylphenol, cadmium, copper, gamma-BHC, and pentachlorophenol.

[i] Liver endpoint is for 4-methylphenol, aldrin, alpha-BHC, beta-BHC, BEHP, butyl benzyl phthalate, chlordane, copper, total DDTs, dieldrin, endrin, endrin aldehyde, gamma-BHC, heptachlor, heptachlor epoxide, hexachlorobenzene, and pentachlorophenol.

[j] Neurological endpoint is for 4-methylphenol, mercury, and total PCBs.

[k] Dermal endpoint is for 4-methylphenol and arsenic.

API – Asian and Pacific Islander
BEHP – bis(2-ethylhexyl) phthalate
BHC – benzene hexachloride
COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
HI – hazard index
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TBT – tributyltin

Table 6-6. Summary of estimated excess cancer risks for direct sediment exposure scenarios

CHEMICAL	NETFISHING		BEACH PLAY RME								CLAMMING		
	RME	CT	AREA 1	AREA 2	AREA 3	AREA 4	AREA 5	AREA 6	AREA 7	AREA 8	TRIBAL – 183 DAYS PER YEAR	TRIBAL RME	7 DAYS PER YEAR
Arsenic	6×10^{-6}	1×10^{-6}	5×10^{-6}	7×10^{-6}	4×10^{-6}	4×10^{-6}	3×10^{-6}	3×10^{-6}	4×10^{-6}	3×10^{-6}	3×10^{-5}	2×10^{-5}	3×10^{-7}
cPAHs[a]	1×10^{-6}	2×10^{-7}	1×10^{-5}	4×10^{-5}	3×10^{-5}	8×10^{-6}	5×10^{-6}	5×10^{-6}	1×10^{-6}	4×10^{-6}	8×10^{-6}	5×10^{-6}	1×10^{-7}
Dioxin and furan TEQ	2×10^{-5}	4×10^{-6}	na	na	na	1×10^{-5}	8×10^{-8}	na	6×10^{-8}	na	2×10^{-4}	1×10^{-4}	8×10^{-7}
PCB TEQ	4×10^{-6}	6×10^{-7}	4×10^{-9}	3×10^{-7}	na	9×10^{-6}	1×10^{-7}	3×10^{-7}	3×10^{-8}	8×10^{-8}	5×10^{-5}	3×10^{-5}	2×10^{-7}
Total PCBs	2×10^{-6}[b]	3×10^{-7}[b]	7×10^{-8}	1×10^{-7}	1×10^{-7}	6×10^{-6}	1×10^{-7}	5×10^{-7}	1×10^{-7}	1×10^{-7}	1×10^{-5}[b]	8×10^{-6}[b]	9×10^{-8}
Subtotal (excluding PCB TEQ)	**3×10^{-5}**	**6×10^{-6}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**9×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**2×10^{-4}**	**1×10^{-4}**	**1×10^{-6}**
Subtotal (excluding total PCBs)	**3×10^{-5}**	**6×10^{-6}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**8×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**3×10^{-4}**	**2×10^{-4}**	**1×10^{-6}**
Tentatively Identified Chemicals (JN-qualified)													
Toxaphene	2×10^{-6}	3×10^{-7}	1×10^{-8}	7×10^{-9}	2×10^{-8}	2×10^{-7}	1×10^{-7}	1×10^{-8}	1×10^{-8}	1×10^{-8}	9×10^{-6}	6×10^{-6}	2×10^{-8}
Total risk (excluding PCB TEQ) across both exposure routes)[c]	**3×10^{-5}**	**5×10^{-6}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**9×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**3×10^{-4}**	**1×10^{-4}**	**1×10^{-6}**
Total risk (excluding total PCBs) across both exposure routes)[c]	**3×10^{-5}**	**6×10^{-6}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**8×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**3×10^{-4}**	**2×10^{-4}**	**1×10^{-6}**

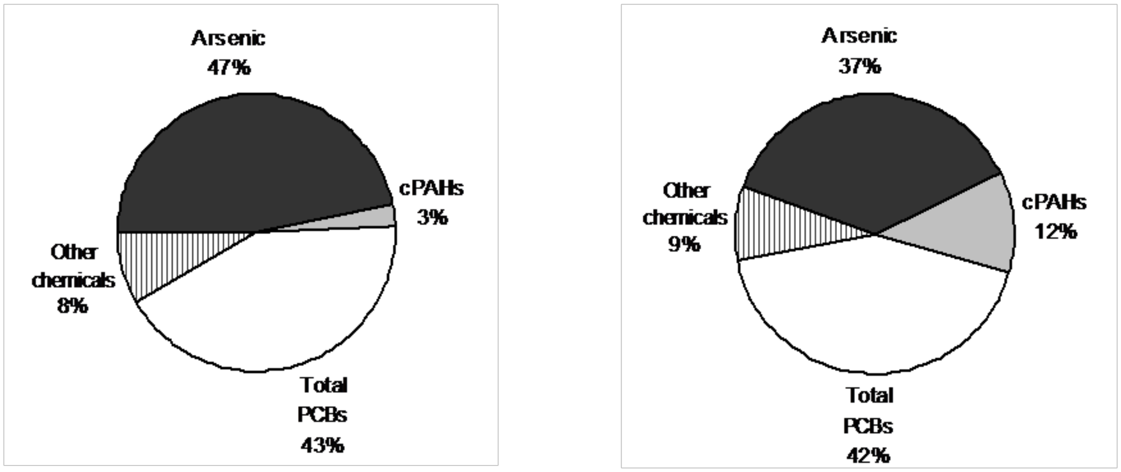
[a] cPAH concentrations are based on benzo(a)pyrene equivalents. Because of the potential for increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (2005e), the risk estimate for beach play RME for cPAHs is based on dose adjustments across the 0-to-6-year age range of children. See Appendix B, Section B.5.1, for more information.

[b] The EPC used for this risk estimate was based on an arithmetic UCL, which is expected to overestimate exposure because of spatially biased sampling. The arithmetic mean was greater than the spatially-weighted mean (developed using Thiessen polygons) by a factor of approximately 5.

[c] Total risk values include the risks associated with all COPCs. However, only those COPCs with an excess cancer risk greater than or equal to 1×10^{-6} for at least one scenario are listed in this table.

COPC – chemical of potential concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
EPC – exposure point concentration
na – not available
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
SWAC – spatially weighted average concentration
TEQ – toxic equivalent

A small number of chemicals were responsible for the majority of the estimated health risks, and the highest risk estimates were associated with seafood consumption. The distributions of cancer risks by chemical are shown in Figure 6-2 for both the adult and child seafood consumption scenarios. The average percent contribution of different chemicals is shown for the adult scenarios because there was little variability in relative chemical contribution among the different scenarios (i.e., adult tribal RME based on Tulalip data, adult API RME, and adult tribal based on Suquamish data). Approximately 85 to 90% of health risks were associated with only two or three chemicals (i.e., arsenic, PCBs, or cPAHs), and almost all the risks in the "other chemicals" category were attributed to tentatively identified pesticides. Cancer risks from cPAHs represented a higher fraction of total cancer risks for children as compared with adults because the risk characterization method used assigns higher cPAH toxicity to children as compared with adults. Note that Figure 6-2 does not show risks for dioxins and furans, which were not analyzed in tissue. The majority of the non-cancer hazards associated with seafood consumption were contributed by total PCBs (> 80% of the total developmental, neurological, and immunological hazard indices). The total PCB HQ for all seafood consumption scenarios, except the adult one-meal-per-month crab scenario, exceeded 1.



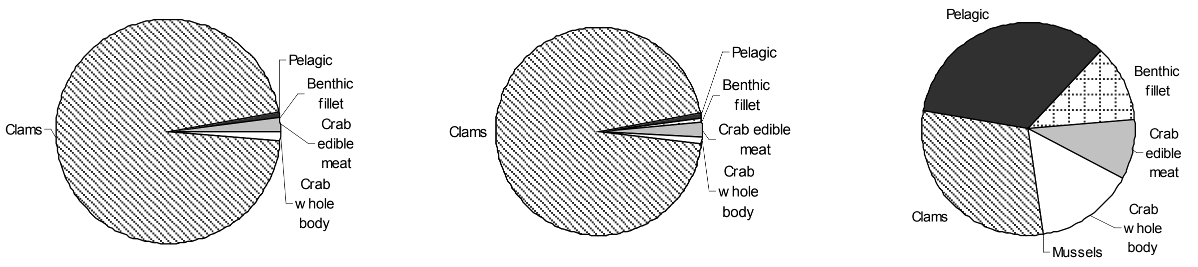
a) Average total cancer risk (excluding PCB TEQ) for adult seafood consumption scenarios

b) Average total cancer risk (excluding PCB TEQ) for child seafood consumption scenarios

Figure 6-2. Seafood consumption scenario risks by chemical

An incremental risk evaluation was performed for arsenic in tissue because arsenic is a an element found in the sediments of central and northern Puget Sound as result of natural sources (see Appendix B, Section B.5.5.1). Tissue samples for several species were collected from background locations and analyzed for arsenic. To estimate the incremental arsenic risk associated with the consumption of LDW seafood, arsenic risk calculated for consumption of seafood from background areas was subtracted from arsenic risk calculated for consumption of seafood from the LDW. Consideration of arsenic background risk did not appreciably affect LDW risk estimates associated with clam consumption. However, the incremental arsenic risk associated with the LDW was negligible for all other seafood categories assessed.

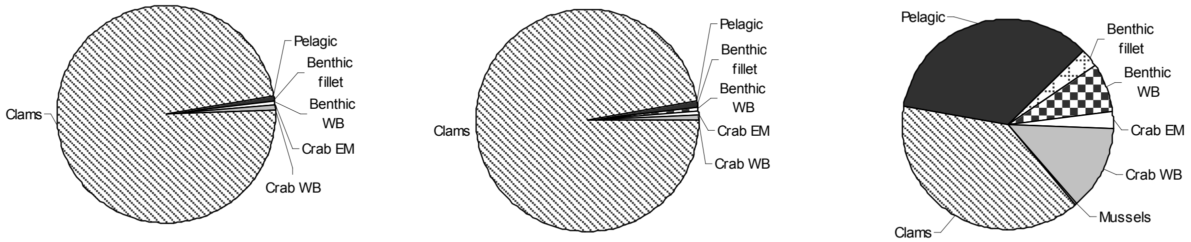
Elevated risk estimates associated with cPAHs and arsenic in seafood were largely attributed to consumption of clams (Figure 6-3). In contrast, risks from total PCBs through the consumption of seafood were more evenly divided among the seafood consumption categories (Figure 6-3, Panels c, f, and i), with pelagic fish and clams as dominant contributors. There were relatively small differences between the three scenarios (i.e., adult tribal RME based on Tulalip data, adult API RME, and adult tribal based on Suquamish data) for arsenic and cPAHs. For total PCB risks, the proportional contributions of all fish compared with all shellfish were similar across the three scenarios. However, within the shellfish group, the types of shellfish contributing the most to total PCB excess cancer risk differed across the three groups, reflecting differences in shellfish consumption patterns across the scenarios.



a) Arsenic cancer risks by seafood category for the adult tribal RME seafood consumption scenario based on Tulalip data

b) cPAH cancer risks by seafood category for the adult tribal RME seafood consumption scenario based on Tulalip data

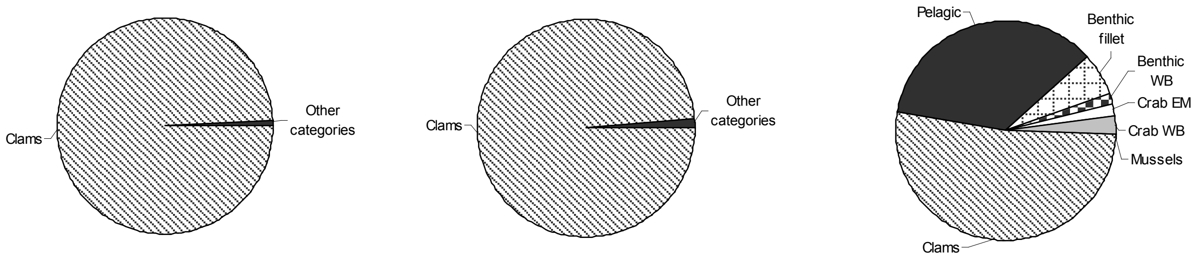
c) Total PCB cancer risks by seafood category for the adult tribal RME seafood consumption scenario based on Tulalip data



d) Arsenic cancer risks by seafood category for the API RME seafood consumption scenario

e) cPAH cancer risks by seafood category for the API RME seafood consumption scenario

f) Total PCB cancer risks by seafood category for the API RME seafood consumption scenario



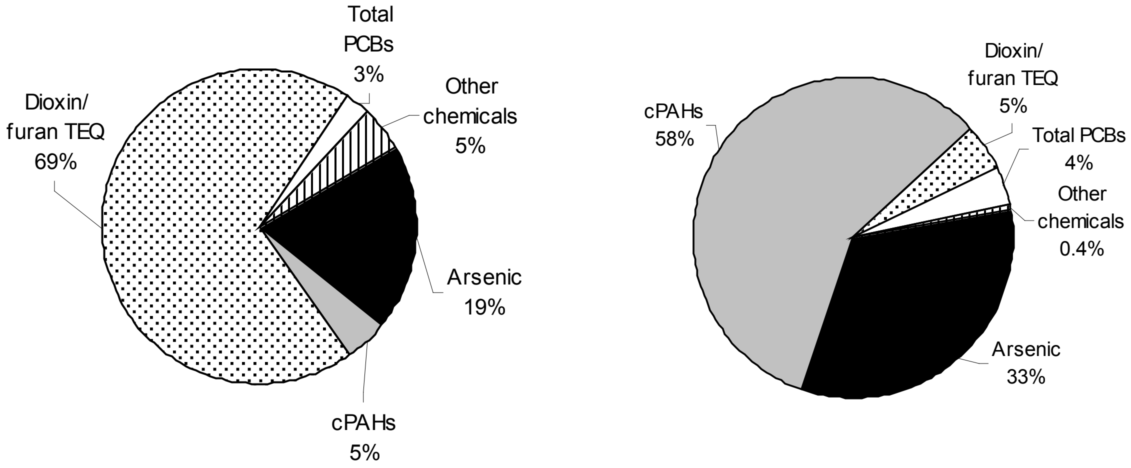
g) Arsenic cancer risks by seafood category for the adult tribal seafood consumption scenario based on Suquamish data

h) cPAH cancer risks by seafood category for the adult tribal seafood consumption scenario based on Suquamish data

i) Total PCB cancer risks by seafood category for the adult tribal seafood consumption scenario based on Suquamish data

Figure 6-3. Excess cancer risks by seafood category for the adult seafood consumption scenarios

The risk contribution by chemical was different for direct sediment exposure scenarios than for seafood consumption exposure scenarios. The direct sediment exposure risk estimates were also quite different between the adult (i.e., netfishing and clamming) and child (i.e., beach play) scenarios. Dioxin and furan risks contributed the majority (average of 69%) of the risks for the adult sediment scenarios, followed by arsenic (average of 19%) (Figure 6-4, Panel a). For the beach play scenarios, cPAHs were responsible for the majority of the estimated risks (average of 58%) and dioxins and furans were much less important to the overall risk estimate (average of 5%), primarily because the highest concentrations of dioxins and furans in the LDW were not in beach play areas, and there were fewer data available, so risks associated with these chemicals could not be evaluated for some areas (Figure 6-4, Panel b). In addition, the risk characterization method used for cPAHs assigns higher cPAH toxicity to children as compared with adults, so cPAH cancer risk estimates for the beach play scenario represented a much higher fraction of the total cancer risk as compared with the adult scenarios. In areas where dioxin and furan data were lacking, estimates of their contribution to risk could not be made.



a) Average total cancer risk estimate for the two netfishing and three clamming scenarios

b) Average total cancer risk estimate for the eight beach play RME scenarios

Figure 6-4. Excess cancer risks by chemical for direct-contact sediment exposure scenarios

In addition to estimating the risks for each exposure scenario separately, risks for multiple scenarios were summed to represent possible exposure of the same individuals to LDW chemicals from different activities, as shown in Table 6-7. The sum of excess cancer risk estimates for each of these four sets of scenarios is the same as the estimates for their seafood consumption components alone after rounding to one significant figure, as recommended by EPA (1989). This analysis demonstrates that the contributions of netfishing, clamming, beach play, and swimming are relatively small in comparison to seafood consumption risk estimates and highlights the significance of the seafood-consumption exposure pathways for all users of the LDW. Summing child beach play and swimming increased the risk estimate only slightly over that of beach play alone. Overall, swimming was the least important exposure pathway.

Table 6-7. Excess cancer risk estimates across related scenarios

ACTIVITY	EXCESS CANCER RISK[a]
Adult Tribal Scenarios	
Adult tribal netfishing RME[b]	3×10^{-5}
Swimming[c]	$< 1 \times 10^{-6}$
Adult tribal RME seafood consumption (Tulalip data)	3×10^{-3}
Total	$\mathbf{3 \times 10^{-3}}$
Child Scenarios[d]	
Beach play RME – Area 2[e]	5×10^{-5}
Swimming[c]	$< 1 \times 10^{-6}$
Subtotal for beach play and swimming	$\mathbf{5 \times 10^{-5}}$
Child tribal RME seafood consumption (Tulalip data)	8×10^{-4}
Total	$\mathbf{9 \times 10^{-4}}$
Adult Low-End Clamming Scenarios	
Clamming – 7 days per year	1×10^{-6}
Swimming[c]	$< 1 \times 10^{-6}$
Clam consumption – one meal per month	2×10^{-4}
Total	$\mathbf{2 \times 10^{-4}}$
Adult Tribal RME Clamming Scenarios	
Tribal clamming RME (120 days per year)	1×10^{-4}
Swimming[c]	$< 1 \times 10^{-6}$
Adult tribal RME seafood consumption based on Tulalip data	3×10^{-3}
Total	$\mathbf{3 \times 10^{-3}}$

[a] All non-swimming risk estimates are presented in Appendix B; total excess cancer risk estimates excluding PCB TEQ were used because these were equal to or higher than total excess cancer risk estimates excluding total PCBs.

[b] Although EPA guidance generally discourages summing risk estimates from multiple RME scenarios, risks for the netfishing RME scenario, rather than the netfishing CT scenario, were added to the seafood consumption RME scenario to account for the fact that RME seafood consumption and RME netfishing may be practiced by tribal members simultaneously.

[c] Adult and child swimming risk estimates as reported by King County for Elliott Bay and the Duwamish River for medium exposure assumptions (12 events per year for adults or children aged 1 to 6) (King County 1999c). Exposure pathways consisted of dermal contact and incidental sediment ingestion of water during swimming. Risks were estimated based on total PCB concentrations of 0.0144 µg/L in the LDW originally modeled by King County (King County 1999c). PCB congener data from samples collected from the LDW by King County in 2005 indicate that this modeled estimate is likely an overestimate of actual total PCB concentrations, which were no greater than 0.0314 µg/L during low-flow sampling conducted in August 2005 (Mickelson and Williston 2006). These results indicate that the risk estimates for the swimming scenario presented by King County in the water quality assessment (King County 1999c) are also likely overestimated.

[d] Child scenarios include the child tribal RME seafood consumption estimate based on 40% of the total adult tribal RME consumption based on Tulalip data, which is considered protective of non-tribal children.

[e] Beach play Area 2 is included because it had the highest risk estimate of the beach play scenarios.

CT – central tendency

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

6.4 UNCERTAINTY ANALYSIS

There are many uncertainties associated with the risk estimates for each exposure scenario in the baseline HHRA. To be health-protective of all members of the general public, the risk estimates presented in the baseline HHRA were intended to avoid underestimation of risks for the RME individual, and thus are likely to overestimate risks for most individuals for the chemicals that were evaluated. Although risk estimates were highest for the seafood consumption scenarios, the uncertainties associated with those risk estimates were also very high. There is considerable uncertainty about the applicability of some of the seafood consumption rates to this HHRA under current uses of the site, particularly for clams, given the quality and quantity of shellfish habitat in the LDW. These risk estimates were intended to provide information to risk managers involved in remedial planning at the site but may not reflect actual risks to people currently consuming LDW seafood.

Another important uncertainty is in the methods used to characterize the excess cancer risks associated with exposures to PCBs. Two methods were used in the baseline HHRA, one method based on total PCB data and the cancer SF for total PCBs and a second method based on dioxin-like PCB congener data and the cancer SF for TCDD. Because both of these methods account for the potential carcinogenic effects of PCBs, the excess cancer risk estimates from these two methods were not summed in estimating cumulative risks in order to avoid double-counting PCB excess cancer risks. Hence, the risk estimates for the two methods were presented separately in the baseline HHRA. Although this approach avoids the double-counting of PCB-related excess cancer risks, it is possible that each method for quantifying PCB excess cancer risks on its own underestimates the overall PCB cancer health risk. The issues associated with assessing risks posed by environmental PCB mixtures, various approaches for addressing double-counting, and quantitative risk estimates derived using these approaches are discussed in detail in the baseline HHRA uncertainty analysis section (Appendix B, Section B.6.3.1).

Dioxins and furans were not analyzed in seafood samples. This lack of data contributes to an underestimation of risk because these chemicals were not included in the risk assessment for the seafood consumption scenarios.

The final risk estimates also reflect uncertainties associated with using data and assumptions from multiple sources; the combined effect of those uncertainties on risk estimates cannot be quantified. However, the risk assessment tended to overestimate risks more than underestimate them, consistent with the health-protective nature of risk assessment. All or any of the uncertainties are relevant to the risk estimates. In spite of these uncertainties, the baseline risk characterization for the LDW site is considered to be health-protective and sufficient to support risk management decisions.

6.5 SUMMARY AND SELECTION OF HUMAN HEALTH RISK DRIVERS

The baseline HHRA identified risk driver chemicals (referred to as indicator hazardous substances in the MTCA program [WAC 173-340-703]). The risk drivers from both the HHRA and the ERA will be the focus of remedial analyses in the FS. The starting point for designating risk drivers was to identify COCs. Chemicals with excess cancer risk estimates greater than 1×10^{-6} or an HQ greater than 1 for any RME exposure scenario were designated COCs. Nineteen chemical were identified as seafood consumption COCs.[120] Eighteen chemicals were identified as seafood consumption COCs based on an exceedance of one or both of these risk thresholds for at least one seafood RME consumption scenario. Dioxins and furans were also included as a COC for seafood consumption, even though no quantitative risk estimates were made. It was assumed that dioxin and furan excess cancer risk estimates for seafood consumption scenarios would have been greater than 1×10^{-6}. Five chemicals were identified as direct sediment exposure COCs[121] based on excess cancer risk estimates greater than 1×10^{-6} for at least one RME scenario. Note that none of the direct sediment contact scenarios had HQs greater than 1; thus, no chemicals were designated as COCs based on non-cancer risk for the direct-contact scenarios.

Organochlorine pesticides were identified as COCs for several seafood consumption and direct sediment scenarios. However, as discussed previously, there may have been analytical interference from PCBs that affected their identification and quantification. Because there is no known source of organochlorine pesticides in the LDW, health risks associated with organochlorine pesticides in the LDW would not be expected to differ from those for other, similar urban or industrial waterway environments.

Chemicals selected as risk drivers were a subset of the COCs. The relative percentage of the total human health risk posed by these COCs was considered in designating a risk driver. Other factors that were also considered included the criteria identified in WAC 173-340-703 for designating an indicator hazardous substance: (a) toxicological characteristics that influence its ability to adversely affect human health or the environment relative to the concentration of the hazardous substance at the site, (b) tendency to persist in the environment, (c) tendency to move into and through environmental media, (d) preliminary natural background concentrations, (e) thoroughness of testing, (f) detection frequency, and (g) chemicals that readily break down into less toxic byproducts (see Appendix B, Section B.7, for more details). Based on this analysis, four risk drivers were identified for both seafood consumption and

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[120] The COCs based on seafood consumption scenarios were PCBs, arsenic, cPAHs, dioxins and furans, aldrin, BEHP, alpha-BHC, beta-BHC, carbazole, total chlordane, total DDTs, dieldrin, gamma-BHC, heptachlor, heptachlor epoxide, hexachlorobenzene, pentachlorophenol, TBT, and vanadium.

[121] The COCs based on direct sediment exposure were PCBs, arsenic, cPAHs, dioxins and furans, and toxaphene.

direct sediment exposure: PCBs, arsenic, cPAHs, and dioxins and furans. The direct-contact risk estimate for dioxins and furans was driven by high concentrations in surface sediments at a small number of sampling locations within the LDW.

COCs not selected as risk drivers in the baseline HHRA will be addressed through focused evaluation in the FS; these chemicals may also be considered in remedial design for specific areas in or near the LDW, and included in the post-remedial monitoring program that is part of the 5-year review that EPA conducts at all Superfund sites where hazardous substances, pollutants, or contaminants remain in place after cleanup is completed.

7 Preliminary Background Concentrations of Risk Driver Chemicals

This section presents concentrations of selected risk driver chemicals (i.e., PCBs, arsenic, cPAHs, and dioxins and furans) in sediment, tissue, and water collected from locations within and outside of the LDW. Understanding background concentrations of these risk driver chemicals is important because it provides a context for both the ecological and human health RBTCs, which are discussed in Section 8. The datasets and summary statistics presented in this section represent information available at the time of the RI; additional data for use in evaluating background are being collected and evaluated. The specific data and summary statistics presented in this section are preliminary; the final background values considered in setting preliminary remediation goals (PRGs) and assessing remedial alternatives will be presented in the FS. EPA and Ecology will select the remediation goals and the final background concentrations to be considered in the derivation of cleanup levels in the Record of Decision (ROD).

Preliminary background concentrations of PCBs, arsenic, cPAHs, and dioxins and furans are presented in this section because some of the risk-based threshold concentrations (RBTCs; see Section 8) for these chemicals are lower than concentrations in areas outside of the LDW. RBTCs for other risk-driver chemicals based on the SQS and CSL appear to be higher than these concentrations and thus are not presented in this section.

MTCA (WAC 173-340-200) defines natural background as concentrations of hazardous substances that are consistently present in an environment that has not been influenced by localized human activities. Thus, under MTCA, natural background values can be defined for man-made compounds (e.g., PCBs) that may not occur naturally. MTCA also discusses area background, which is a term used to represent those concentrations that are consistently present in the environment in the vicinity of the site as a result of human activities unrelated to releases from the site.

CERCLA uses different definitions for background. Under CERCLA guidance, anthropogenic background refers to natural and human-made substances that are present in the environment as a result of human activities, but that are not specifically related to CERCLA releases at the site (EPA 2002d). Natural background refers to substances that are naturally present in the environment in forms that have not been influenced by human activity (e.g., naturally occurring metals). Under both the CERCLA and MTCA programs, anthropogenic or area background concentrations are generally higher than natural background concentrations.

This section is intended to provide initial background information to provide a preliminary context for the site characterization data and RBTCs presented elsewhere in the RI report. Some additional datasets that are relevant to background exist; this compilation is not exhaustive. The exclusion of a dataset from these discussions does not necessarily exclude those data from being considered in the future as background evaluations continue, and some data included in this summary may be eliminated in future background evaluations. Additional sampling studies are underway and will be providing new datasets relevant to LDW background determinations. The presentation of information provided in this section is not intended to be interpreted as the development of background-based PRGs or cleanup levels in the RI.

Background-related information may be relevant for multiple chemicals and for multiple environmental media. As noted above, there are also several terms for "background" (e.g., natural, anthropogenic, area), which are based on different concepts and usually require different datasets for characterizing each type of background concentration. For any single chemical, environmental medium, and type of background, repeated sampling may produce different sets of results. Therefore, final background determinations by EPA and Ecology may be based on a (statistical) distribution of values and not a single value. Statistical evaluations of background datasets can become complex. For the purpose of providing preliminary information in this section, the focus is on presenting relatively simple summary statistics. Further background assessments and the development of background-based PRGs and cleanup levels will involve detailed statistical evaluations, as appropriate.

An evaluation of background requires an assessment of the quality of the various potential datasets before deciding how to evaluate the data statistically. The general term "representativeness" can be used to reflect multiple considerations, such as sampling and analysis protocols, possible changes over time, spatial differences, the potential influence of other sources, and the specific relevance of the data, as collected and analyzed, to the LDW. The selection of datasets to serve as the basis for background determinations for the LDW will consider such representativeness issues but is not attempted in this section. If more than one dataset is selected, background determinations will consider combining the data versus performing separate evaluations of each dataset and then assessing the multiple results (e.g., as independent "lines of evidence"). These detailed considerations and dataset combinations are not addressed for this RI summary but will be addressed in future background analyses. Consistent with the intent of this section to provide only an initial context related to background, the general approach is simple and descriptive rather than complex and interpretive.

Section 7.1 presents sediment data for selected risk driver chemicals from samples collected from areas that are not believed to have been influenced by localized human activities. These data are relevant for consideration of natural background concentrations.

Section 7.2 discusses concentrations of chemicals in sediments collected upstream of the LDW and in sediments that have been deposited in the Upper Turning Basin of the LDW. Area and anthropogenic background concentrations in sediment have not been formally defined for the LDW. Sediment transported from upstream of the LDW is likely to significantly influence the future quality and characteristics of sediment within the LDW (see Section 3). Therefore, multiple sediment datasets are presented in Section 7.2 along with a discussion of associated physical properties that may affect their chemistry. Section 7.2 also discusses upstream water quality data and estimates chemical concentrations in suspended sediment in the water column. These data provide a range of relevant chemical concentrations that will be considered in the FS in the assessment of PRGs and the evaluation of remedial alternatives.

Section 7.3 provides additional context for the data presented in Sections 7.1 and 7.2 by presenting urban sediment data from Seattle and other Puget Sound locations.

Tissue data from Puget Sound are presented in Section 7.4. Most of the available tissue data for total PCBs from Puget Sound have been characterized as either near-urban or non-urban (West et al. 2001). Available arsenic data were categorized according to whether or not the sampling location was likely to have been impacted by emissions from the former Asarco smelter located in Ruston, Washington. Tissue data are presented to show the general range of concentrations in areas outside the influence of site-related activities.

7.1 SEDIMENT DATASETS TO BE CONSIDERED FOR NATURAL BACKGROUND

This section provides a summary of available sediment data from reference areas in the Puget Sound region. Summary statistics are provided for each reference area or specific study; combined preliminary natural background concentrations were not derived. These data are considered to be representative of non-urban, non-localized concentrations in the Puget Sound region that exist as a result of natural processes and/or the large-scale distribution of chemicals from anthropogenic sources.

7.1.1 Data sources

Ten areas in Puget Sound have been identified as reference areas by the Battelle Marine Research Laboratory (Battelle) (1986) or WSDOH (1995) (Table 7-1; Maps 7-1, 7-2, and 7-3). Sediment data from these 10 reference areas were obtained from four sources: 1) a study by NOAA and Ecology on sediment quality in Puget Sound conducted from 1997 to 1999 (NOAA and Ecology 1999, 2000, 2002), 2) Washington State's SEDQUAL database (Ecology 2004c), 3) a Battelle survey of eight bays (1986), and 4) Ecology's Environmental Information Management (EIM) System database (Ecology 2007c). Data were obtained for total PCBs, arsenic, and cPAHs from this search. Data were processed to the extent possible using data management procedures described in Appendix E, although the data from these datasets were generally used as reported in the above sources without further data quality reviews. In addition, sampling and analytical methods used to produce these datasets varied from study to study and potentially from year to year within a given study. Thus, while these data provide a general indication of chemical concentrations in these reference areas, they should not be viewed as a single dataset with consistent methodology.

Table 7-1. Reference areas identified for consideration of natural background concentrations of total PCBs, arsenic, and cPAHs

Reference Area	Sources Used to Identify Reference Areas
Carr Inlet	WSDOH (1995)
Case Inlet	Battelle (1986)
Dabob Bay	WSDOH (1995); Battelle (1986)
Discovery Bay	WSDOH (1995)
Rich Passage	WSDOH (1995)
Samish Bay	WSDOH (1995); Battelle (1986)
Saratoga Passage	WSDOH (1995)
Sequim Bay	WSDOH (1995); Battelle (1986)
West Beach	WSDOH (1995)
Wollochet Bay	WSDOH (1995)

Battelle – Battelle Marine Research Laboratory
cPAH – carcinogenic polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
WSDOH – Washington State Department of Health

In addition to the data from the reference areas identified above, additional data were obtained from the following studies:

- **Rayonier, Inc., RI –** Samples were collected and analyzed for dioxin, furan, and PCB congeners from two areas outside the influence of localized human activities (Dungeness Bay and Freshwater Bay) as part of the RI at the Rayonier, Inc., mill site near Port Angeles, Washington (Map 7-1) (Malcolm Pirnie 2007b).
- **Puget Sound 2008 Survey –** Samples were collected and analyzed from 14 reference areas throughout Puget Sound at established reference sites and locations distant from known sources of contamination (Map 7-4) (Liebman 2008). Five samples were collected from each reference area for a total of 70 samples. Samples were analyzed for PCBs (Aroclors and PCB congeners), arsenic, cPAHs, and dioxins and furans, in addition to other analytes.

In the following sections, data from individual studies are summarized separately. The Puget Sound 2008 survey data are summarized as a combined dataset from multiple areas because sampling and analytical methods were consistent in the 14 areas sampled throughout Puget Sound.

The DMMP has compiled a set of dioxin and furan data representing background in Puget Sound, but these data were not available for inclusion in the RI (USACE 2009). The DMMP's data compilation includes the Puget Sound 2008 survey data as well as additional data from the Anderson-Ketron open-water disposal site, Ecology's EIM, and a reference sample associated with a recent dredge project sediment characterization.

7.1.2 Arsenic surface sediment datasets to be considered for natural background

Arsenic was frequently detected in samples from all 10 reference areas in Puget Sound and was detected in all of the samples from the Puget Sound 2008 survey (Figure 7-1). The arsenic concentrations included in the datasets ranged from 0.5 mg/kg dw in Carr Inlet to 27 mg/kg dw in Sequim Bay (Table 7-2). Detection frequencies in individual areas ranged from 48 to 100%. Sampling locations are shown on Maps 7-2 and 7-3. Percent fines ranged from 1 to 98.5% in the Puget Sound 2008 survey, with a mean of 47.4%.

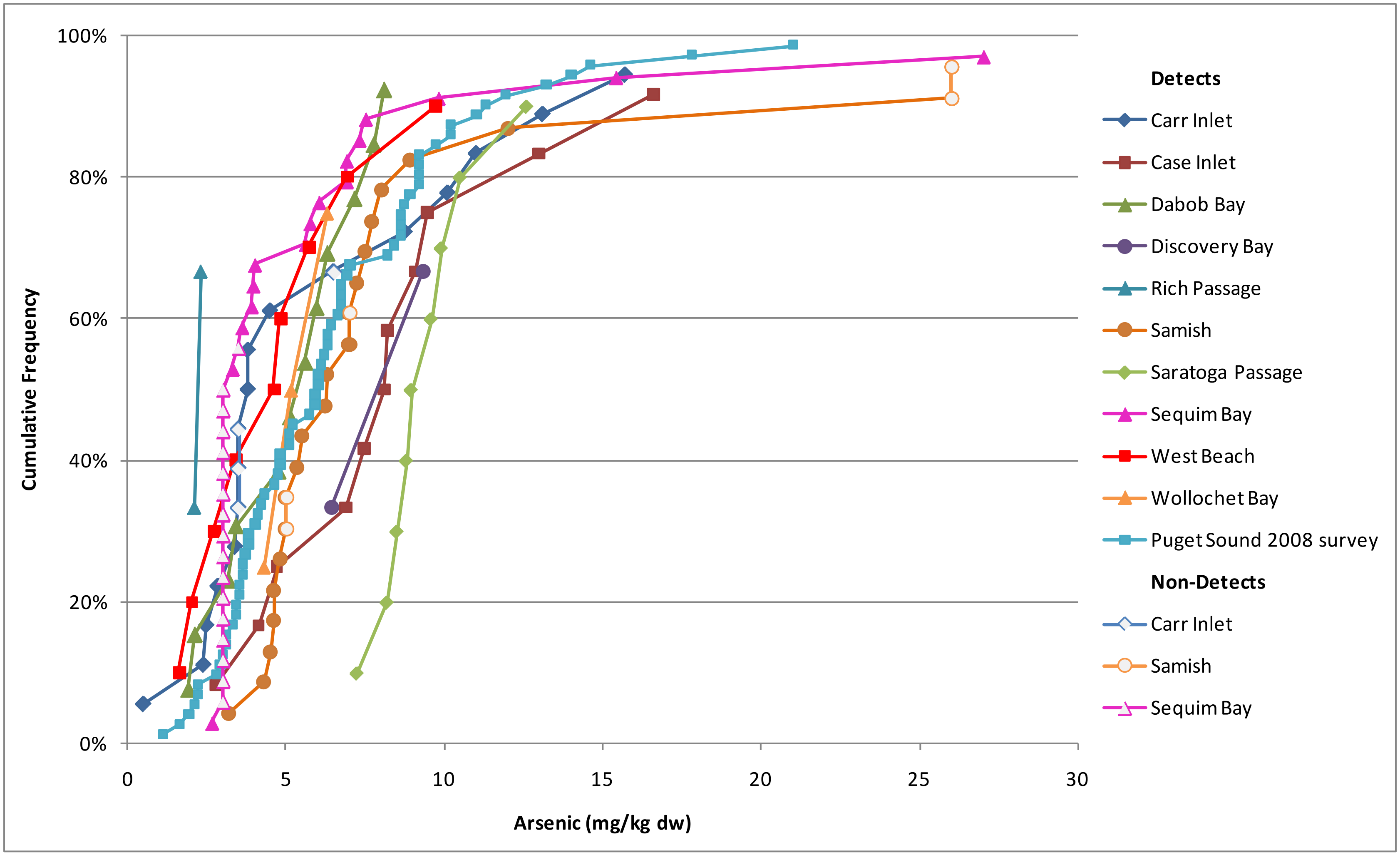


Figure 7-1. Cumulative frequency of arsenic concentrations in surface sediment collected from Puget Sound reference areas and the 2008 survey

Table 7-2. Summary of arsenic datasets to be considered for natural background

Sampling Location	Sampling Period	Detection Frequency	Concentration (mg/kg dw)								UCL Type
			Minimum Detect	Maximum Detect	Mean Detect	Minimum RL	Maximum RL	50th Percentile[a]	90th Percentile[a]	95% UCL[b]	
Reference Areas											
Carr Inlet	1972 – 2000	13/17	0.500	15.7	6.34	7.0	13.0	3.8	13.6	7.59	95% KM (t) UCL
Case Inlet	1972 – 1999	11/11	2.8	16.6	8.2	na	na	8.1	15.9	10.4	95% Student's-t UCL
Dabob Bay	1995 – 1999	12/12	1.90	8.1	5.11	na	na	5.4	8.0	6.2	95% Student's-t UCL
Discovery Bay	1986 – 1994	2/2	6.43	9.3	7.87	na	na	7.9	9.3	9.3	maximum detect
Rich Passage	1998	2/2	2.10	2.3	2.2	na	na	2.2	2.3	2.3	maximum detect
Samish	1989	17/22	3.20	12.0	6.34	10	52	6.3	12.0[c]	7.16	95% KM (t) UCL
Saratoga Passage	1972 – 1997	9/9	7.2	13	9.36	na	na	9.0	12.6	10.3	95% Student's-t UCL
Sequim Bay	1994 – 1997	16/33	2.65	27.0	7.47	6.0	7.0	3.0	8.9	7.2	95% KM (% bootstrap) UCL
West Beach	1986 – 1990	9/9	1.6	9.7	4.6	na	na	4.6	9.7	6.2	95% Student's-t UCL
Wollochet Bay	1982 – 1996	3/3	4.30	6.3	5.25	na	na	5.2	6.3	6.3	maximum detect
Puget Sound 2008 Survey											
Puget Sound	2008	70/70	1.1	21	6.5	na	na	6.0	11	7.3	95% approximate gamma UCL

Note: Multiple datasets have been collected over time, there were no consistent temporal trends in the data.

[a] Percentiles were calculated using detected values and one-half the RL for samples in which arsenic was not detected. Percentiles are non-parametric percentiles calculated using methods presented in the *National Institute of Standards and Technology Handbook of Statistical Methods* (NIST and SEMATECH 2006).

[b] UCLs were calculated using ProUCL 4 statistical software (EPA 2007e), which accounts for non-detect values in the calculation of UCLs.

[c] Maximum detected value presented; the 90th percentile value was higher than the maximum detect because of elevated RLs.

dw – dry weight
KM – Kaplan-Meier
na – not applicable
RL – reporting limit
t – Student's t distribution
UCL – upper confidence limit on the mean

7.1.3 PCB surface sediment datasets to be considered for natural background

Total PCB concentrations were calculated either by summing detected PCB Aroclors or by summing detected PCB congeners. The analytical method used for PCB Aroclors had a much higher RL than the analytical method used for PCB congeners. Thus, PCBs as Aroclors were detected in only 11 of 140 samples that were analyzed using the Aroclor method, while at least one of the 209 PCB congeners was detected in every sample analyzed using the PCB congener method, resulting in 100% detection (Figure 7-2; Table 7-3). In the 70 samples collected for the Puget Sound 2008 survey, the calculated mean concentration of total PCBs as Aroclor sums was 11 µg/kg dw (based on a mean of the total PCB concentration in the six samples with at least one detected Aroclor), compared with 1.2 µg/kg dw for the mean concentration as PCB congener sums (based on a mean of the total PCB concentration in all 70 samples with at least one detected PCB congener). Reference locations with PCB sediment data are presented on Maps 7-1 and 7-2. TOC content in the 70 samples from the Puget Sound 2008 survey ranged from 0.2 to 4.0%, with a mean of 1.3%. When the total PCB data from the Puget Sound 2008 survey were normalized for organic carbon content, concentrations ranged from 0.0056 to 0.85 mg/kg OC, with a mean of 0.11 mg/kg OC (Table 7-3).

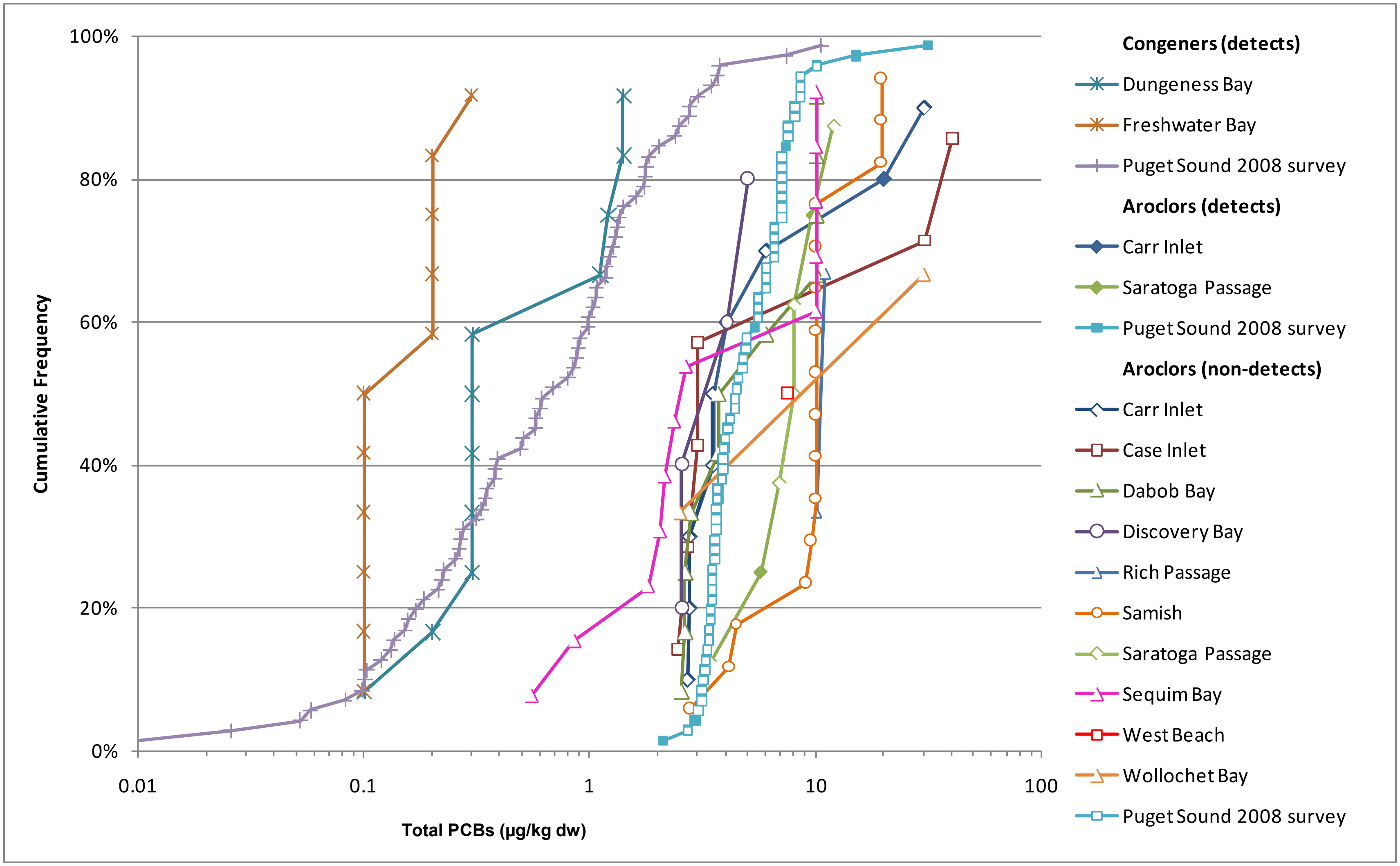


Figure 7-2. **Cumulative frequency of total PCB concentrations in surface sediment collected from Puget Sound reference areas and the 2008 survey**

Table 7-3. Summary of total PCB datasets to be considered for natural background

				Concentration[a]									
Sampling Location	**Sampling Period**	**Analtye**	**Unit**	**Detection Frequency**	**Minimum Detect**	**Maximum Detect**	**Mean Detect**	**Minimum Non-Detect[b]**	**Maximum Non-Detect[b]**	**50th Percentile[c]**	**90th Percentile[c]**	**95% UCL[d]**	**UCL Type**
Reference Areas													
Carr Inlet	1984 – 2000	PCB Aroclors	µg/kg dw	2/9	4.0	19.9	11.5	5.4	60	3.5	19.9[e]	19.9	maximum detect
Case Inlet	1999	PCB Aroclors	µg/kg dw	0/6	na	na	na	4.9	80	nc	nc	nc	nc
Dabob Bay	1995 – 1999	PCB Aroclors	µg/kg dw	0/11	na	na	na	5.1	20	nc	nc	nc	nc
Discovery Bay	1994 – 2002	PCB Aroclors	µg/kg dw	0/4	na	na	na	5.1	10	nc	nc	nc	nc
Dungeness Bay	2006	PCB congeners	µg/kg dw	11/11	0.2	1.4	0.63	na	na	0.3	1.4	1.4	maximum detect
Freshwater Bay	2006	PCB congeners	µg/kg dw	11/11	0.1	0.3	0.15	na	na	0.1	0.2	0.19	95% Modified-t UCL
Rich Passage	1998	PCB Aroclors	µg/kg dw	0/2	na	na	na	20	22	nc	nc	nc	nc
Samish	2000 – 2004	PCB Aroclors	µg/kg dw	0/16	na	na	na	5.5	39	nc	nc	nc	nc
Saratoga Passage	1996 – 1997	PCB Aroclors	µg/kg dw	3/7	5.7	9.7	7.4	6.8	24	8.0	9.7	9.7	maximum detect
Sequim Bay	1994 – 2002	PCB Aroclors	µg/kg dw	0/12	na	na	na	1.1	20	nc	nc	nc	nc
West Beach	1994	PCB Aroclors	µg/kg dw	0/1	na	na	na	15	15	nc	nc	nc	nc
Wollochet Bay	1996 – 1999	PCB Aroclors	µg/kg dw	0/2	na	na	na	5	60	nc	nc	nc	nc
Puget Sound 2008 Survey													
Puget Sound	2008	PCB Aroclors	µg/kg dw	6/70	2.1	31	11	5.4	20	4.4	8.0	6.5	95% KM (percentile bootstrap) UCL

Sampling Location	Sampling Period	Analtye	Unit	Concentration[a]									UCL Type
				Detection Frequency	Minimum Detect	Maximum Detect	Mean Detect	Minimum Non-Detect[b]	Maximum Non-Detect[b]	50th Percentile[c]	90th Percentile[c]	95% UCL[d]	
Puget Sound	2008	PCB Aroclors	mg/kg OC	5/59[f]	0.21	3.0	0.99	0.36	1.97	0.39	0.63	3.0	maximum detect
Puget Sound	2008	PCB congeners	µg/kg dw	70/70	0.010	11	1.2	na	na	0.65	2.8	1.5	95% approximate gamma UCL
Puget Sound	2008	PCB congeners	mg/kg OC	59/59[f]	0.0056	0.85	0.11	na	na	0.054	0.21	0.16	95% H-UCL

Note: Multiple datasets have been collected over time; there were no consistent temporal trends in the data.

[a] For PCB Aroclors, the total PCB concentration represents the sum of detected concentrations of nine individual PCB Aroclors for a given sample. For samples in which none of the individual Aroclors were detected, the maximum RL for an individual PCB Aroclor in that sample is used as the concentration. For PCB congeners, the total PCB concentration represents the sum of the detected PCB congener concentrations for a given sample.

[b] Non-detected concentrations for samples in which none of the individual PCB Aroclors were detected were calculated as described in Footnote a.

[c] Percentiles are non-parametric percentiles calculated using methods presented in the *National Institute of Standards and Technology Handbook of Statistical Methods* (NIST and SEMATECH 2006). Percentiles were calculated using both detected and non-detected total PCB concentrations (see Footnote a).

[d] UCLs were calculated using ProUCL 4 statistical software (EPA 2007e), which accounts for non-detect values in the calculation of UCLs.

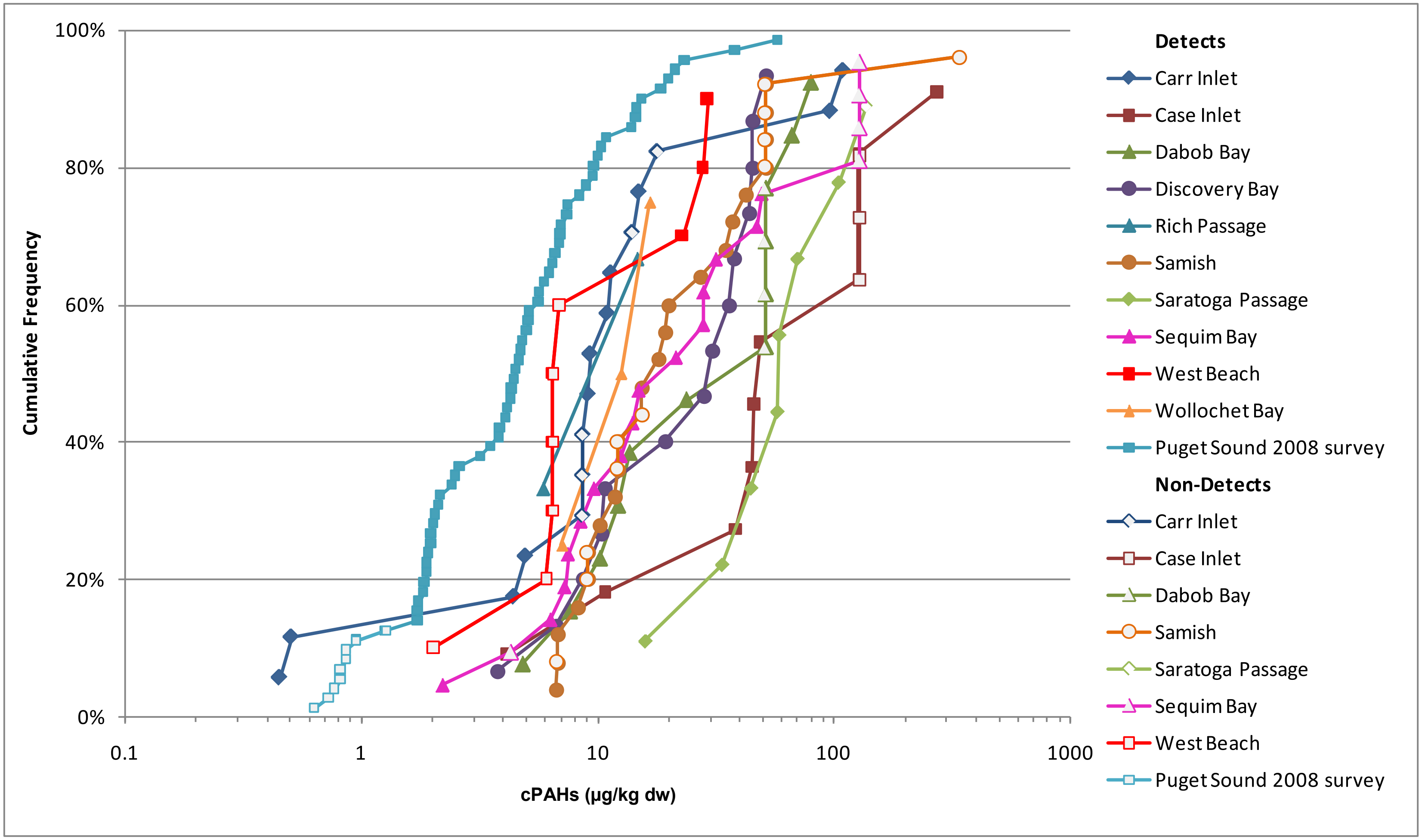
[e] Maximum detected value presented; the 90th percentile value was higher than the maximum detected value because of elevated RLs.

[f] OC-normalized concentrations were not calculated for 11 samples that had TOC contents ≤ 0.5% and ≥ 4%.

dw – dry weight
H – Land's H statistic
KM – Kaplan Meier
na – not available
nc – not calculated
PCB – polychlorinated biphenyl
RL – reporting limit
TOC – total organic carbon
UCL – upper confidence limit on the mean

7.1.4 cPAH surface sediment datasets to be considered for natural background

At least one cPAH compound was detected in the majority of samples from the 10 reference areas in Puget Sound and the Puget Sound 2008 survey (Figure 7-3). The cPAH concentrations ranged from 0.45 µg/kg dw in Carr Inlet to 274 µg/kg dw in Case Inlet (Table 7-4). Detection frequencies in individual areas ranged from 33 to 100%, although the detection frequency for individual cPAH compounds was sometimes as low as 0%. For samples from the Puget Sound 2008 survey, OC-normalized cPAH concentrations ranged from 0.11 to 2.8 mg/kg OC, with a mean concentration of 0.56 mg/kg OC (Table 7-4). Sampling locations for the cPAH samples are shown on Maps 7-3 and 7-4.



Note: Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E.

Figure 7-3. Cumulative frequency of cPAH concentrations in surface sediment collected from Puget Sound reference areas and the 2008 survey

Table 7-4. Summary of cPAH datasets to be considered for natural background

Sampling Location	Sampling Period	Concentration (µg/kg dw)[a]									UCL Type
		Detection Frequency	Minimum Detect	Maximum Detect	Mean Detect	Minimum Non-Detect[b]	Maximum Non-Detect[b]	50th Percentile[c]	90th Percentile[c]	95% UCL[d]	
Reference Areas											
Carr Inlet	1984 – 2000	11/16	0.450	108	24.5	17.2	35.4	9.1	99.2	33.9	95% KM (BCA) UCL
Case Inlet	1992 – 1999	8/11	4.14	274	59.9	256	256	46.0	244.7	154.5	95% KM (Chebyshev) UCL
Dabob Bay	1995 – 1999	8/12	4.78	79.2	27.1	102	102	37.3	75.2	43.1	95% KM (BCA) UCL
Discovery Bay	1994 – 2003	14/14	3.75	51.5	26.9	na	na	29.3	48.5	34.9	95% Student's-t UCL
Rich Passage	1998	2/2	5.84	14.7	10.3	na	na	10.3	14.7	14.7	maximum detect
Samish	1989	13/24	6.63	41.9	19.8	13.4	102	16.7	41.9[e]	21.6	95% KM (t) UCL
Saratoga Passage	1996 – 1997	7/8	15.9	105	55	272	272	58	105[e]	75.3	95% KM (t) UCL
Sequim Bay	1994 – 2002	15/20	2.20	49.4	19.2	8.5	256	18.1	49.4[e]	24.7	95% KM (percentile bootstrap) UCL
West Beach	1986 – 1990	3/9	22.7	29.2	26.6	4.0	13.7	6.4	29.2	29.2	maximum detect
Wollochet Bay	1982 – 1996	3/3	7.00	16.6	12	na	na	12.5	16.6	16.6	maximum detect
Puget Sound 2008 Survey											
Puget Sound	2008	61/70	1.7	58	7.9	1.3	2.5	4.5	15	9.2	95% KM (BCA) UCL
Puget Sound 2008 Survey (mg/kg OC)											
Puget Sound	2008	53/59	0.11	2.8	0.56	0.11	2.8	0.29	1.2	0.81	95% KM (Chebyshev) UCL

Note: Multiple datasets have been collected over time; there were no consistent temporal trends in the data.

[a] Total cPAHs were calculated by summing the products of individual PAH concentrations and compound-specific PEFs for the seven individual cPAH compounds (benzo(a)anthracene, benzo(b)fluoranthene, benzo(a)pyrene, benzo(k) fluoranthene, chrysene, dibenzo(a,h)anthracene, and indeno(1,2,3-cd)pyrene), as discussed in Appendix E. Total cPAHs were considered detected if one or more of the individual cPAH compounds were detected. For non-detected compounds, one-half the RL was multiplied by the PEF when calculating cPAH concentrations.

[b] Non-detected values for samples in which none of the individual cPAH compounds were detected were calculated as described in Footnote a.

[c] Percentiles are non-parametric percentiles calculated using methods presented in the *National Institute of Standards and Technology Handbook of Statistical Methods* (NIST/SEMATEC 2009). For samples in which none of the cPAHs were detected, one-half the calculated cPAH concentration was used (see Footnote a).

[d] UCLs were calculated using ProUCL 4 statistical software (EPA 2007e), which accounts for non-detect values in the calculation of UCLs.

[e] The 90th percentile value was higher than the maximum detected value because of elevated RLs.

BCA – bias-corrected accelerated
cPAH – carcinogenic polycyclic aromatic hydrocarbon
KM – Kaplan Meier
na – not applicable
PEF – potency equivalency factor
RL – reporting limit
TEQ – toxic equivalent
UCL – upper confidence limit on the mean

7.1.5 Dioxins and furans sediment datasets to be considered for natural background

The Puget Sound 2008 survey provides a robust reference dataset for dioxins and furans; prior to this survey relatively few reference data were available for dioxins and furans. At least one dioxin or furan congener was detected in every sample collected as part of the 2008 survey and in the other two Puget Sound locations sampled (Maps 7-1 and 7-2). Among all of the datasets, dioxin and furan TEQs ranged from 0.067 ng/kg dw in a sample collected from Freshwater Bay to 12 ng/kg dw in a sample collected from the South Sound (Figure 7-4; Table 7-5). For samples from the Puget Sound 2008 survey, OC-normalized dioxin and furan TEQs ranged from 0.028 to 0.30 µg/kg OC, with a mean concentration of 0.10 µg/kg OC (Table 7-5).

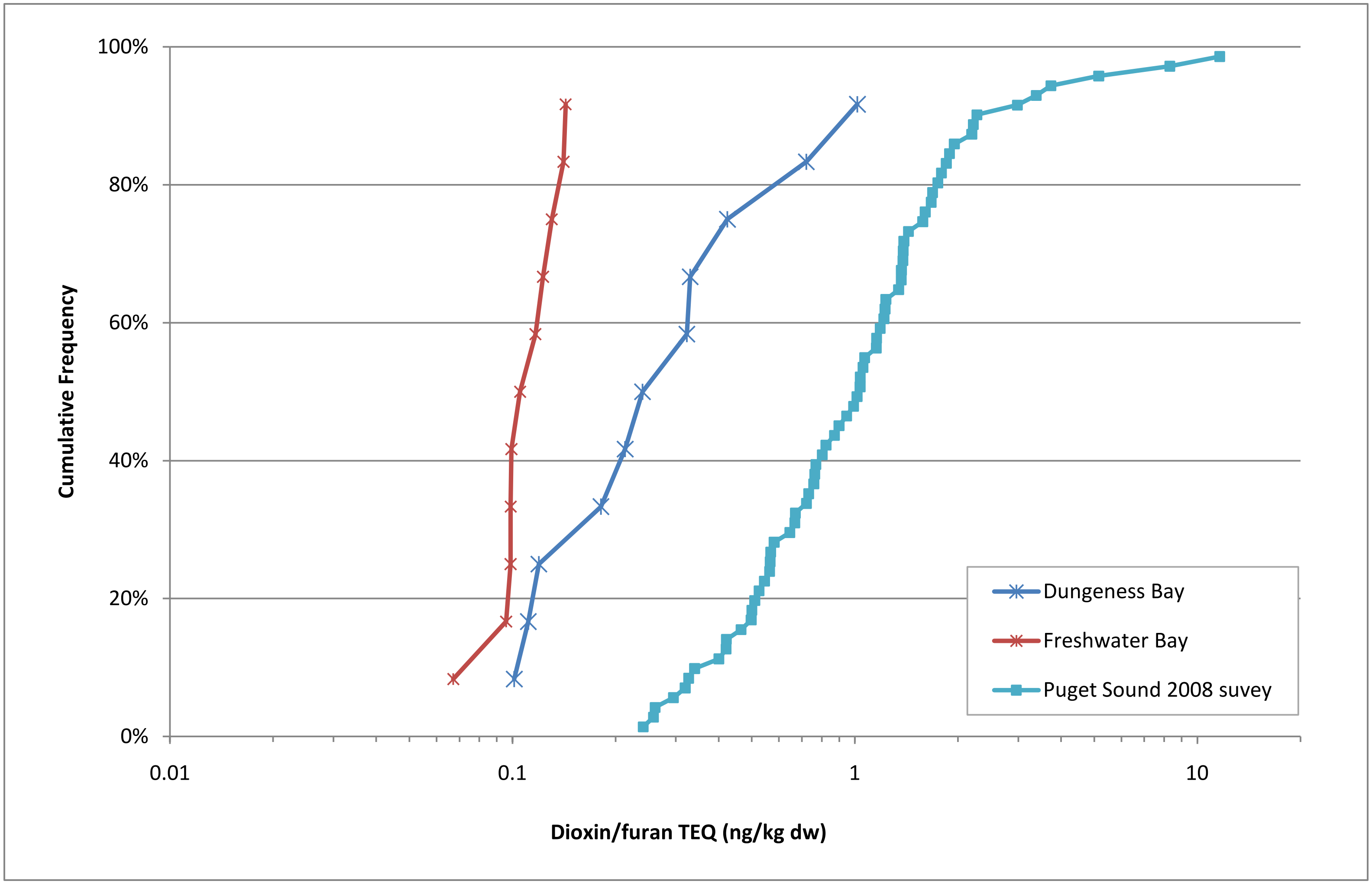


Figure 7-4. Cumulative frequency of dioxin and furan TEQs in surface sediment collected from two Puget Sound locations and as part of the 2008 survey

Table 7-5. Summary of dioxin and furan TEQ datasets to be considered for natural background

SAMPLING LOCATION	SAMPLING PERIOD	CONCENTRATION (ng/kg dw)[a]							UCL TYPE
		DETECTION FREQUENCY	MINIMUM DETECT	MAXIMUM DETECT	MEAN DETECT	50TH PERCENTILE[b]	90TH PERCENTILE[b]	95% UCL[c]	
Dungeness Bay	2006	11/11	0.101	1.02	0.344	0.24	0.721	0.542	95% Approximate gamma UCL
Freshwater Bay	2006	11/11	0.067	0.143	0.111	0.105	0.141	0.123	95% Student's-t UCL
Puget Sound	2008	70/70	0.24	12	1.4	1.0	2.3	1.6	95% H-UCL
TOC-normalized dioxin and furan TEQ data (µg/kg OC)									
Puget Sound	2008	59/59	0.028	0.30	0.10	0.088	0.19	0.12	95% H-UCL

[a] Dioxin and furan TEQs were calculated by summing the products of individual congener concentrations and congener-specific TEFs, as discussed in Appendix E. A dioxin and furan TEQ value was considered detected if one or more of the congeners were detected. For non-detected congeners, the TEF was multiplied by one-half the RL.

[b] Percentiles are non-parametric percentiles calculated using methods presented in the *National Institute of Standards and Technology Handbook of Statistical Methods* (NIST/SEMATEC 2009).

[c] UCLs were calculated using ProUCL 4 statistical software (EPA 2007e), which accounts for non-detect values in the calculation of UCLs.

dw – dry weight
H – Land's H statistic
na – not applicable
nc – not calculated
RL – reporting limit
TEF – toxic equivalency factor
TEQ – toxic equivalent
UCL – upper confidence limit on the mean

7.2 UPSTREAM AND UPPER TURNING BASIN DATA

This section presents concentrations of PCBs, arsenic, cPAHs, and dioxins and furans in surface sediment samples that were collected upstream of the LDW, as well as in subsurface sediment samples that were collected in the Upper Turning Basin of the LDW. Various data were considered as separate lines of evidence. The following three types of data are presented in this section:

- Surface sediment data collected upstream of RM 5.0
- Sediment core data collected between RM 4.0 and RM 4.75 by USACE for dredged material characterization
- Estimated chemical concentrations in upstream suspended solids based on Green/Duwamish River water chemistry and total suspended solids data

The types of data listed above were collected for various purposes; thus, each line of evidence may have uncertainties and biases with respect to characterizing chemical concentrations in sediment likely to deposit within the LDW. Therefore, all lines of evidence will be evaluated as part of the overall background assessment in the FS, wherein the uncertainties and biases for each of the lines of evidence will be discussed and evaluated in detail.

Section 7.2.1 provides a brief overview of sediment transport processes to provide additional context for the different lines of evidence. The data sources and methods for evaluation of each of these datasets are presented in Sections 7.2.2 through 7.2.4.

7.2.1 Review of sediment transport processes and sediment physical properties

As summarized in Section 3, extensive analysis of sediment transport processes has been conducted in recent years, primarily through the use of the STM. One important conclusion reached through the application of the STM is that approximately 99% of the total external sediment load to the LDW comes from the Green River, upstream of the LDW. About 50% of this total load settles in the LDW. The remaining load, comprised mostly of finer silt-clay particles, is transported out of the LDW because these particles have slower settling velocities and thus travel greater distances than coarser particles prior to deposition. Approximately 10% of the fine silt-clay particles that enter the LDW system, either from upstream or lateral sources, are deposited in the LDW. In contrast, 76, 96, and > 99% of the other particle classes (medium-coarse silt, fine sand, and medium-coarse sand, respectively) from upstream or lateral sources are deposited in the LDW.

Twenty-four percent of the total sediment load entering the upstream end of the STM study area (defined as RM 4.75) is bedload,[122] which primarily settles in the Upper

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[122] Bedload is that portion of sediment that can be transported by intermittent contact with the streambed by rolling, sliding, and bouncing.

Turning Basin (RM 4.5 to RM 4.75) (QEA 2008). The remainder of the total sediment load entering the STM study area (76%) is suspended material. This material generally consists of finer fractions (i.e., clay, silt, and fine sand) with lower settling velocities compared with bedload sand, and therefore, is generally transported greater distances in the LDW. These finer fractions may have higher chemical concentrations than the coarser bedload (i.e., medium and coarse sand) because of the affinity of chemicals (such as hydrophobic PCBs, cPAHs, and dioxins and furans) to sorb to finer sediments with high surface area-to-volume ratios and higher TOC (Hedges and Keil 1995). Potential differences in chemical concentrations between larger-size-fraction bedload and smaller-size-fraction suspended materials may also be mirrored by differences in chemical concentrations among the various size fractions of the suspended materials themselves. As noted in Section 3, much of the finest particulate matter from upstream may be carried through the LDW without depositing to LDW sediments, and chemical concentrations may increase as particle size decreases. These issues related to representativeness of potential background datasets will be evaluated in further background analyses and determinations but are not assessed in this section. All representativeness issues and sources of potential bias in candidate background datasets will be discussed in detail in the FS.

As a result of the sediment transport processes described above, sediments sampled in various locations tend to have differing physical properties. Table 7-6 summarizes the TOC content and percent fines (sum of silt and clay fractions) of the upstream sediment data (i.e., surface sediment data from RM 5.0 to RM 7.0), subsurface sediment data from the navigation channel between RM 4.0 and RM 4.75, and surface sediment data from the LDW up to RM 4.0. The subsurface sediment chemistry data were further subdivided to reflect longitudinal differences between the Upper Turning Basin and sediments collected closer to RM 4.0.

The TOC and percent fines values for available surface sediment samples upstream of RM 5.0 were much lower than those values for surface sediments below RM 4.0. These results are consistent with the observation that suspended solids, which primarily consist of fine particles with relatively higher TOC content (compared with sand) are transported from upstream regions (above RM 5.0) throughout the LDW. The subsurface sediment data from RM 4.0 to RM 4.4 were similar to data from surface sediment in RM 0.0 to RM 4.0 (Table 7-6). In contrast, the data close to and within the Upper Turning Basin more closely resembles the surface sediment data from upstream of RM 5.0. These results are consistent with the observation that coarser bedload fractions from upstream regions are preferentially deposited within and near the Upper Turning Basin compared with more downstream locations.

Table 7-6. Total organic carbon and percent fines data for upstream sediment datasets compared with LDW surface sediment from RM 0.0 to RM 4.0

PARAMETER	SURFACE SEDIMENT (RM 0.0 to RM 4.0)	SUBSURFACE SEDIMENT (RM 4.0 TO RM 4.4)[a]	SUBSURFACE SEDIMENT (RM 4.4 TO RM 4.75)[a]	SURFACE SEDIMENT (RM 5.0 TO RM 7.0)
Total Organic Carbon (%)				
Average	1.97	1.94	1.26	0.83
Minimum	0.05	0.60	0.03	0.068
Maximum	11.5	2.80	3.10	2.31
10th percentile	0.91	1.41	0.44	0.08
90th percentile	2.89	2.60	2.74	1.93
Percent Fines				
Average	52.8	58.2	17.0	24.1
Minimum	1.6	23.0	2.6	0.01
Maximum	100	78.0	36.7	65.0
10th percentile	13.5	49.0	6.0	0.01
90th percentile	85.8	69.0	34.2	57.0

[a] The subsurface sediment samples collected from the navigation channel between RM 4.0 and 4.75 generally represent composite samples from sediment cores that start at the mudline and therefore include sediment from the surface interval (i.e., the top 10 cm).

LDW – Lower Duwamish Waterway

RM – river mile

7.2.2 Upstream surface sediment data – RM 5.0 to RM 7.0

Data were compiled for total PCBs, arsenic, cPAHs, and dioxins and furans in both suspended matter and bedded sediments upstream of the LDW (RM 5.0 to RM 7.0); sampling locations are shown on Map 7-5.[123] These data were evaluated to assess the quality of sediment potentially being transported into the LDW from the Green/Duwamish River system. Table 7-7 presents a statistical summary of the data. Dioxins and furans and cPAHs were calculated as TEQs using methods described in Appendix E, Sections E.3.6 (dioxins and furans) and E.3.7 (cPAHs).

Relatively few upstream surface sediment locations were sampled for dioxins and furans (4 locations) compared with 37, 24, and 16 locations for total PCBs, arsenic, and cPAHs, respectively (Table 7-8). Mean and 90th percentile concentrations were 23 and 40 µg/kg dw for total PCBs, 6.8 and 11 mg/kg dw for arsenic, and 55 and 135 µg/kg dw for cPAH, respectively (Table 7-7). The mean dioxin and furan TEQ was 2.0 ng/kg

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[123] Because of the small number of dioxin and furan samples collected between RM 5.0 and RM 7.0 (three), one sample collected from Springbrook Creek (which enters the Duwamish River through the Black River at approximately RM 10.2) was included in the dioxin and furan dataset.

dw (Table 7-7); percentiles were not calculated for dioxins and furans because of the small sample size.

Ecology conducted a study investigating chemical concentrations in bedded sediments between RM 4.9 and RM 6.5, and the data resulting from that study will also be considered in the FS when they are available. The Ecology study was conducted to reduce the uncertainty in the characterization of chemical conditions in upstream sediments, which is currently based on a relatively small number of samples relative to the number of sediment samples in the LDW study area, and to assess whether there are any point sources in this upstream area. The existing upstream surface sediment data were collected for various purposes and may not provide an unbiased representation of chemical concentrations in upstream areas.

Table 7-7. Summary statistics for upstream datasets and USACE cores in the Upper Turning Basin of the LDW

CHEMICAL AND DATA TYPE	NO. OF SAMPLES	DETECTION FREQUENCY (%)	CONCENTRATION					PERCENTILE[b]		95% UCL[c]	UCL TYPE[c]
			MINIMUM DETECT	MAXIMUM DETECT	MINIMUM NON-DETECT[a]	MAXIMUM NON-DETECT[a]	MEAN[b]	50TH	90TH		
Total PCBs (µg/kg dw)[d]											
Upstream surface sediment[e]	37	49	1.6	140	0.6	40	23	19	40	21	95% KM (BCA) UCL
RM 4.0 to RM 4.75 – USACE cores[f]	37	81	10.8	300	20	52	56	38	86	96	95% KM (BCA) UCL
Upstream suspended solids[g, h]	22	100	2.8	367	na	na	50	21	107	82	95% Approximate Gamma UCL
Total PCBs (mg/kg OC)											
Upstream surface sediment[e, i]	20	55	0.23	10.5	0.1	3.7	2.1	1.0	3.7	5.6	97.5% KM (Chebyshev) UCL
RM 4.0 to RM 4.75 – USACE cores[f, i]	35	86	0.86	15.8	1.1	6.8	3.3	2.1	6.2	5.4	95% KM (Chebyshev) UCL
Upstream suspended solids[g, i, j]	18	100	0.06	2.7	na	na	0.5	0.2	1.3	1.5	97.5%Chebyshev UCL
Arsenic (mg/kg dw)											
Upstream surface sediment[e]	24	100	3.3	22	na	na	6.8	5.4	11	8.0	95% Approximate Gamma UCL
RM 4.0 to RM 4.75 – USACE cores[f]	35	97	3.0	15	9.4	9.4	7.6	6.5	13	8.4	95% KM (BCA) UCL
Upstream suspended solids[g, h]	100	100	0.50	133	na	na	37	29	73	47	95% Chebyshev UCL
cPAHs (µg/kg dw)											
Upstream surface sediment[e]	16	44	36	260	9	18	55	18	135	100	95% KM (t) UCL
RM 4.0 to RM 4.75 – USACE cores[f, k]	36	97	14.6	270	14	14	117	107	201	137	95% KM (t) UCL
Upstream suspended solids [g,h,]	18	67	29	408	22	244	151	74	354	269	95% KM (Chebshev) UCL
cPAHs (mg/kg OC)											
Upstream surface sediment[e, i]	7	86	1.8	20	1.8	1.8	7.1	2.5	18	20	95% KM (Chebshev) UCL
RM 4.0 to RM 4.75 – USACE cores[f, i, k]	34	97	0.64	21	0.4	0.4	7.3	6.6	12	8.7	95% KM (BCA) UCL
Upstream suspended solids [g,i, j]	15	60	0.77	6.8	1.2	6.5	2.8	1.9	5.5	3.0	95% KM (t) UCL
Dioxins and furan TEQ (ng/kg dw)											
Upstream surface sediment[e]	4	100	1.1	2.6	na	na	2.0	1.7	nc	nc	nc

CHEMICAL AND DATA TYPE	NO. OF SAMPLES	DETECTION FREQUENCY (%)	CONCENTRATION MINIMUM DETECT	CONCENTRATION MAXIMUM DETECT	CONCENTRATION MINIMUM NON-DETECT[a]	CONCENTRATION MAXIMUM NON-DETECT[a]	CONCENTRATION MEAN[b]	PERCENTILE[b] 50TH	PERCENTILE[b] 90TH	95% UCL[c]	UCL TYPE[c]
Dioxin and furan TEQ (µg/kg OC)											
Upstream surface sediment[e, i]	2	100	0.1	0.2	na	na	0.2	nc	nc	nc	nc

Source: AECOM (Rodriguez Lattuada 2010)

[a] For arsenic, non-detect values are equal to the full RL. For total PCBs, cPAHs, and dioxin and furans, non-detects indicate that no individual components of the calculated total or TEQ were detected. For total PCBs, the total was calculated as the sum of detected values for the individual components, and the non-detect value is the maximum RL for an individual Aroclor. For cPAH and dioxin and furan TEQs, non-detect values were calculated using the full the RL for non-detected components multiplied by the PEF or TEF, respectively.

[b] The mean and percentile concentrations were calculated using statistical software ProUCL 4.00.04 (EPA 2007e), with non-detects set at full reporting limit.

[c] UCLs were calculated using ProUCL 4 statistical software (EPA 2007e), which accounts for non-detect values in the calculation of UCLs. Selected UCL is the recommended UCL value by ProUCL.

[d] Seventeen samples were analyzed for PCB Aroclors and twenty samples were analyzed using the NOAA method described in Section 4.1.3.3.

[e] Upstream surface sediment samples from 1994 to 2006.

[f] USACE core (RM 4.0 to 4.75) samples from 1990 to 2003.

[g] Upstream suspended solids samples from 2001 to 2008 (King County whole water samples). King County surface water data normalized to solid fractions by dividing by the TSS in the individual sample. All detected arsenic concentrations associated with TSS were calculated as the difference between whole-water (i.e., unfiltered) and filtered sample data.

[h] Concentrations of chemicals on suspended solids were estimated as described in Section 7.2.4 (in units of µg/kg TSS for total PCBs and cPAHs and mg/kg TSS for arsenic).

[i] Some samples were not OC-normalized because the TOC was ≤ 0.5 or ≥ 4%. Therefore, the sample size for the OC-normalized data is smaller than the sample size for the dry weight data.

[j] Data were normalized to the POC content in water. The POC was calculated as the difference between the TOC and the DOC. Only samples with both types of data and where the TOC was greater than the DOC were used. Therefore, the sample size for the OC-normalized data is smaller than the sample size for the TSS normalized data.

[k] One sample concentration (1,052 µg/kg dw) was excluded from the cPAH dataset because it was an outlier (as determined by outlier tools in ProUCL software).

BCA – bias-corrected accelerated
cPAH – carcinogenic polycyclic aromatic hydrocarbon
DOC – dissolved organic carbon
dw – dry weight
KM – Kaplan-Meier
LDW – Lower Duwamish Waterway
na – not applicable
nc – not calculated
OC – organic carbon
PCB – polychlorinated biphenyl
PEF – potency equivalency factor
POC – particulate organic carbon
RL – reporting limit
RM – river mile
TEF – toxic equivalency factor
TEQ – toxic equivalent
TOC – total organic carbon
TSS – total suspended solids
UCL – upper confidence limit on the mean
USACE – US Army Corps of Engineers

Table 7-8. Surface sediment sampling events conducted upstream of the LDW

			Number of Samples (number of detects)				
Sampling Event (Event Code)	**Year Conducted**	**River Mile**	**PCBs**	**Arsenic**	**cPAHs**	**Dioxins and Furans**	**Source**
LDW RI: surface sediment sampling for chemical analyses and toxicity testing (LDWRI-SurfaceSediment Round2)	2005	5.1 – 5.8	6 (0)	6 (6)	6 (4)	0	Windward (2005d)
LDW RI: surface sediment sampling for chemical analyses and toxicity testing (LDWRI-SurfaceSediment Background)	2005	5.3 (PCBs) 6.1 – 7.0 (arsenic) 5.3 and 10.2 (dioxins and furans)[a]	1 (0)	8 (8)	0	2 (2)	Windward (2005i)
EPA site inspection: Lower Duwamish River (EPA SI)	1998	5.3 – 5.5	5 (0)	5 (5)	5 (0)	2 (2)	Weston (1998)
Duwamish Waterway Phase 1 site characterization (Boeing SiteChar; upstream reference samples)	1997	6.1	3 (0)	3 (3)	3 (2)	0	Exponent (1998)
Duwamish Waterway sediment characterization study (NOAA SiteChar)	1997	5.2 – 6.0	20 (18)	0	0	0	NOAA (1997, 1998)
Norfolk CSO sediment cleanup study – Phase 1 (Norfolk-cleanup1)	1994	5.4 – 5.5	2 (0)	2 (2)	2 (1)	0	King County (1996)
Total numbers of samples			**37 (18)**	**24 (24)**	**16 (7)**	**4 (4)**	

[a] This sample was collected in Springbrook Creek, which enters the Duwamish River at approximately RM 10.2.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
CSO – combined sewer overflow
EPA – US Environmental Protection Agency
LDW – Lower Duwamish Waterway
NOAA – National Oceanographic and Atmospheric Administration
PCB – polychlorinated biphenyl
RI – remedial investigation
SI – site investigation

7.2.3 USACE dredged material

The Seattle District USACE has been characterizing sediments in the Upper Turning Basin and in the navigation channel immediately downstream of the Upper Turning Basin as part of their on-going navigation maintenance dredging responsibilities. The area between RM 4.3 and RM 4.75 is dredged approximately every 2 years, while the section between RM 4.0 and RM 4.3 is dredged approximately every 4 years. The material dredged from this area over the past 15 years has been deemed acceptable for open-water disposal by the USACE under the DMMP program. Data from this portion of the navigation channel provides another line of evidence for characterizing the potential contribution of upstream sediment to the LDW because they represent relatively recent material deposited from upstream of the LDW. There are uncertainties in the use of this dataset for characterizing this contribution, primarily related to the partitioning between coarse and fine-grained material in this region, and because of the unknown contribution from sources within the LDW (Table 7-6). The suitability of these data for characterizing background will be discussed in the FS.

USACE provided chemical and physical data for subsurface sediment samples collected between RM 4.0 and RM 4.75 from 1990 to 2003 (Fox 2007) based on a database query of the Dredged Analysis Information System (DAIS). Subsurface sediment samples were also collected in this area in 2008 and 2009, although data were not available for inclusion in this evaluation. For the purposes of dredged sediment characterization, the USACE has been compositing sediment cores for characterization both vertically (generally 0-to-4-ft intervals but occasionally as deeper intervals, up to 10 ft), and in some cases, horizontally (compositing two or more cores collected across the basin or navigation channel). Figures 7-5, 7-6, and 7-7 present the total PCB, arsenic, and cPAH concentrations, respectively, as a function of time relative to dredging events. The sampling locations are shown on Map 7-6.

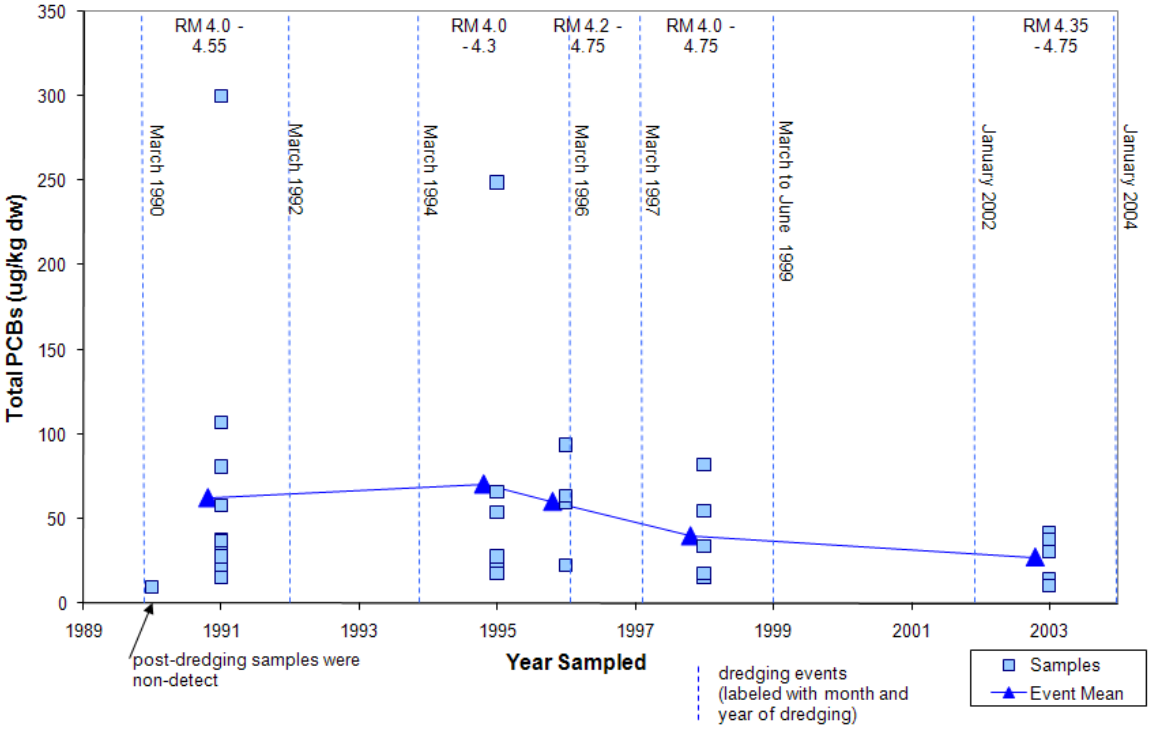


Figure 7-5. Total PCB concentrations in cores collected for dredged material characterization from RM 4.0 to RM 4.75 over time

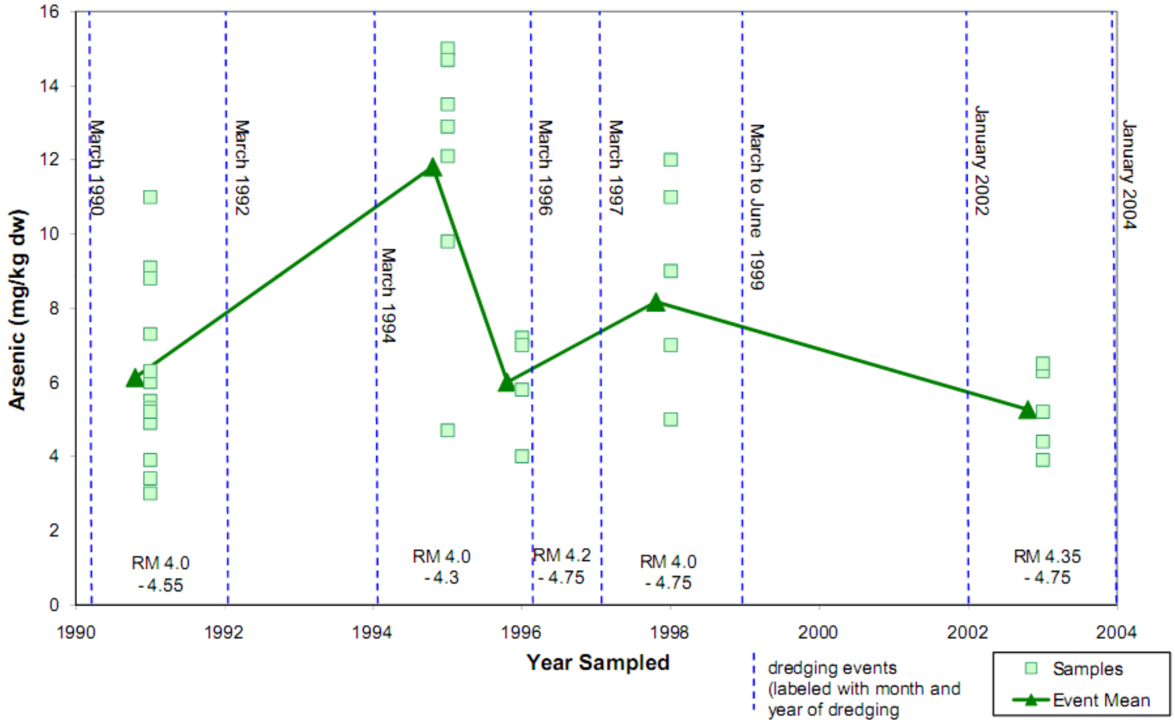
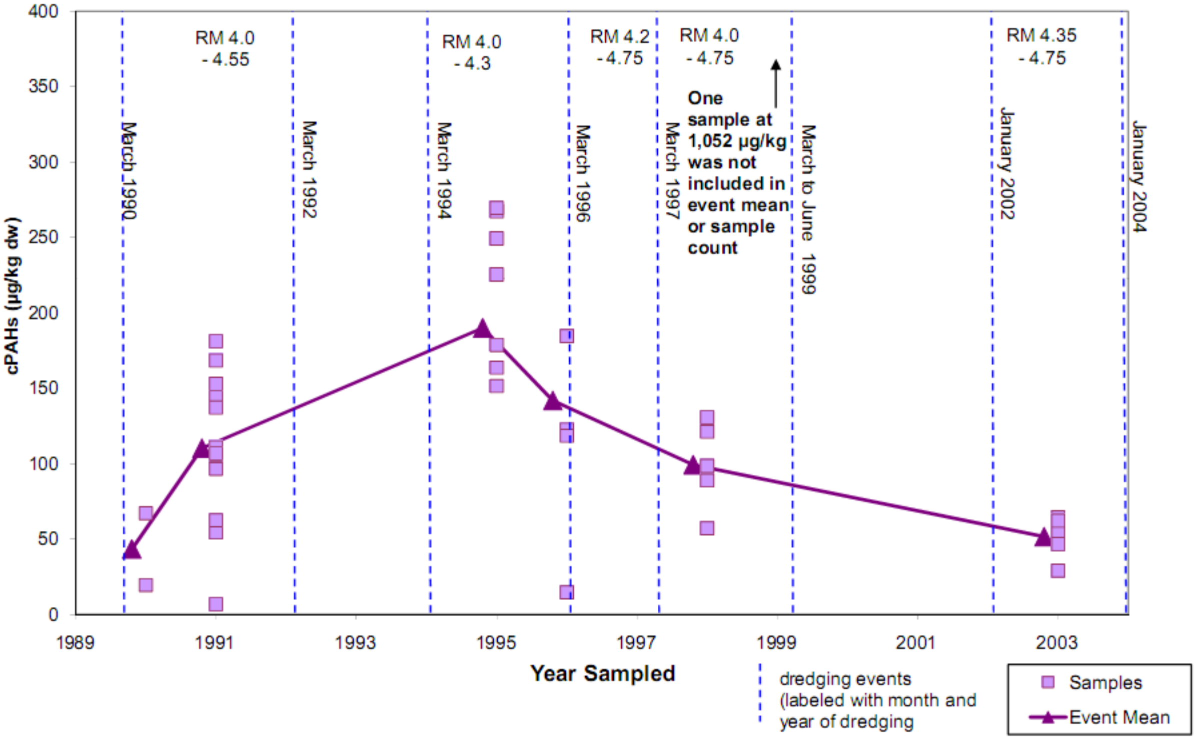


Figure 7-6. Arsenic concentrations in cores collected for dredged material characterization from RM 4.0 to RM 4.75 over time



Note: Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

Figure 7-7. cPAH concentrations in cores collected for dredged material characterization from RM 4.0 to RM 4.75 over time

The mean and 90th percentile concentrations for data collected between 1990 and 2003 were 56 and 86 µg/kg dw for total PCBs, 7.6 and 13 mg/kg dw for arsenic, and 117 and 201 mg/kg dw for cPAH (Table 7-7). One sample concentration (1,052 µg/kg dw) was excluded from the cPAH dataset because it was an outlier (as determined by outlier tools in ProUCL software). Additional analysis of potential outliers will be conducted in the FS.

7.2.4 Water quality data

As discussed in Section 7.2.1, solids may be transported through bedload processes or in suspension in the water column. Suspended solids typically have higher proportions of finer particles, such as silt and clay, compared with material transported by bedload processes, which typically consists of sands. By using chemical data for whole water (i.e., unfiltered) samples and the TSS concentrations for those same samples, an estimate of the chemical concentration in the suspended solids can be generated.

There are a number of uncertainties and potential biases associated with these estimates to characterize chemical concentrations on upstream solids that settle in the LDW. These uncertainties include:

- Greater temporal variation in water quality vs. sediment quality (grab samples vs. integrated sampling technique)
- Small number of storm events sampled with moderate to high TSS, when much of the sediment transport occurs
- Assumptions regarding the extent of partitioning of PCBs and cPAHs in unfiltered water samples to particles
- The relevance of these data with respect to deposition of sediment within the LDW (e.g., potential influence of particle size fraction on chemical concentrations, and differential deposition and transport throughout the LDW)

These factors will be discussed and evaluated further in the FS. In addition, Ecology is currently conducting a study investigating the chemical concentrations associated with suspended solids in high-volume water samples upstream of the LDW (Ecology 2008b). The Ecology data will also be considered in the FS when they are available.

King County water quality data from the lower Green/Duwamish Rivers were used to estimate the concentrations of total PCBs (as a sum of PCB congeners), arsenic, and cPAHs associated with suspended solids in samples collected upstream of the LDW. The King County sampling events for which data are available for this analysis are summarized in Table 7-9. Two stations have been sampled by King County in the lower Green and Duwamish Rivers upstream of the LDW: the Duwamish River at E Marginal Way S (RM 6.3) and the Green River at Fort Dent (~RM 10.5 to RM 11.0; Map 7-7). Whole-water concentrations of PCBs and cPAHs were normalized to the sample-specific TSS concentration and expressed as µg/kg TSS. Solids-associated arsenic concentrations were calculated as the difference between total and filtered sample results, normalized to the sample-specific TSS and expressed as mg/kg TSS.

Table 7-9. Available water quality data from the Lower Green/Duwamish Rivers upstream of the LDW

ANALYTE	SAMPLING LOCATION	TYPE OF DATA	SAMPLING YEARS	STUDY AND REFERENCE
PCB congeners[a]	Green River – Fort Dent	grab samples; total (unfiltered)	2005	Duwamish River/Elliott Bay/Green River water column PCB congener survey (Mickelson and Williston 2006)
			2007 – 2008	unpublished data from King County; (2002b, 2007f)
	Duwamish River at E Marginal Way S		2007	unpublished data from King County (2002b, 2007f)
Arsenic	Green River – Fort Dent	grab samples; total (unfiltered) and dissolved (filtered)	2001 – 2003	Green-Duwamish Watershed water quality assessment (Herrera 2004b, 2005).
			2004 –2006	unpublished data from King County (King County 2002b, 2007f)
cPAHs[a]	Green River – Fort Dent	grab samples; total (unfiltered)	2008	unpublished data from King County (King County 2002b, 2007f)
	Duwamish River at E Marginal Way S			

Note: Each sample summarized above was also analyzed for TSS.

[a] King County analyzed unfiltered water samples collected from the Green River location in 2001 to 2003 for PAHs and PCBs as Aroclors, but these compounds were not detected in any samples (Herrera 2004b, 2005).

cPAH – carcinogenic polycyclic aromatic hydrocarbon

LDW – Lower Duwamish Waterway

PCB – polychlorinated biphenyl

TSS – total suspended solids

The following subsections present the results of King County water quality data for PCBs, arsenic, and cPAHs.

7.2.4.1 PCB congeners

King County collected unfiltered surface water samples from the Green River at Fort Dent from 2001 to 2003 for analyses of total PCBs (as Aroclors) (Map 7-7). PCBs were not detected in any of these samples at detection limits ranging from 0.04 to 0.06 µg/L (Herrera 2004b, 2005).

Thus, in order to achieve lower detection limits, King County analyzed a total of 21 upstream water samples in 2005 and 2007/2008 for all 209 PCB congeners using EPA Method 1668. Unfiltered surface water samples were collected from the same location at Fort Dent four times in 2005 (August to December), nine times in 2007 (March to December), and once in 2008 (June) and analyzed for PCB congeners and TSS (Mickelson and Williston 2006) (Table 7-10). Two of the 2007 samples and the June

2008 sample were collected as part of King County's wet weather event streams sampling program, and thus were collected during storm events. Surface water samples were also collected from the Duwamish River at the E Marginal Way S Bridge at S 155th Street (RM 6.3) seven times in 2007 (March to December) and analyzed for PCB congeners and TSS (Map 7-7) (Axys 2008).

Table 7-10. Total PCB and TSS concentrations in upstream surface water samples and estimated total PCB concentrations associated with suspended solids

		Surface Water				Suspended Solids (estimated)	
		Total PCB (µg/L)[b]		TSS (mg/L)		Total PCBs (µg/kg TSS)	
Data Type	Daily Average Flow (cfs)[a]	Green River at Fort Dent	Duwamish River at E Marginal Way S	Green River at Fort Dent	Duwamish River at E Marginal Way S	Green River at Fort Dent	Duwamish River at E Marginal Way S
Individual Events							
August 22, 2005	277	0.000222	no data	5.2	no data	43	no data
September 26, 2005	378	0.0007839	no data	8.0	no data	98	no data
November 28, 2005	1,060	0.000442[c]	no data	3.6[c]	no data	110[d]	no data
December 19, 2005	550	0.00004084	no data	2.0 U	no data	20	no data
March 7, 2007	1,680	0.0001286	0.0005144	14.7	9.6	8.7	53.6
April 4, 2007	1,620	0.0001012[c]	0.00009948	35.8[c]	10	2.8[d]	9.9
May 9, 2007	1,460	0.00003845	0.0000762[c]	5.8	2.1[c]	6.6	36[d]
May 21, 2007 – storm[e]	985	0.001543	no data	4.2	no data	370	no data
June 6, 2007	840	0.0000491[c]	0.0001460	4.3[c]	3.5	11[d]	42
October 10, 2007	941	0.0000542[c]	0.0000711	7.8[c]	9.3	7.0[d]	7.6
November 7, 2007	539	0.00004046	0.0000603[c]	1.8	1.5[c]	23	41[d]
December 3, 2007 – storm[e]	2,550	0.002395[c]	no data	71.1[c]	no data	34.0[d]	no data
December 12, 2007	1,200	0.000057[c]	0.0000699	10.8[c]	7.4	5.3[d]	9.4
June 3, 2008 – storm[e]	2,190	0.000240	no data	21.8	no data	9.3	no data
August 20, 2008 – storm[e]	321	0.0015	no data	9.24	no data	162	no data
Summary Statistics							
Number of samples	na	22		22		22	
Detection frequency	na	22 / 22		22 / 22		22 / 22	
Mean	1,162	0.000392		11		50[f]	
Minimum	277	0.0000385		1.5		2.8	
Maximum	2,550	0.002395		71.1		367	
RL or range of RLs	na	na		na		na	
25th Percentile	547	0.000056		3.5		8.2[f]	
50th Percentile	1023	0.000100		6.9		21[f]	
75th Percentile	1635	0.000460		13.6		42[f]	

		SURFACE WATER				SUSPENDED SOLIDS (estimated)	
		TOTAL PCB (µg/L)[b]		TSS (mg/L)		TOTAL PCBS (µg/kg TSS)	
DATA TYPE	DAILY AVERAGE FLOW (cfs)[a]	GREEN RIVER AT FORT DENT	DUWAMISH RIVER AT E MARGINAL WAY S	GREEN RIVER AT FORT DENT	DUWAMISH RIVER AT E MARGINAL WAY S	GREEN RIVER AT FORT DENT	DUWAMISH RIVER AT E MARGINAL WAY S
90th Percentile	2,370	0.001530		25.3		107[f]	

[a] Flow data are from the USGS station 12113000 at Auburn, Washington.

[b] Total PCB concentrations were calculated as the sum of detected PCB congeners, which is consistent with the totaling method used for total PCBs based on Aroclors.

[c] Field replicates were collected for these samples; the average total PCB and TSS concentrations shown represent averages of the original and field replicates.

[d] Field replicates were collected for these samples. The average TSS-normalized total PCB concentrations were first calculated separately for the original and the field replicate, and then the two concentrations were averaged.

[e] A storm was a targeted event with at least 0.25-in. rainfall over 24 hours, with a minimum of 24 hours antecedent dry. The other sampling events were conducted during routine monthly monitoring and may or may not correspond to storm events.

[f] The mean and percentile concentrations were calculated using statistical software ProUCL 4.00.04 (EPA 2007e), with non-detects set at full reporting limit.

cfs – cubic feet per second

dw – dry weight

na – not applicable

PCB – polychlorinated biphenyl

RL – reporting limit

TSS – total suspended solids

Summary statistics were calculated for total PCB concentrations (sum of detected PCB congeners) in surface water and estimated total PCB concentrations in suspended solids.[124] Total PCB concentrations in surface water samples ranged from 0.00003845 to 0.002395 µg/L, with mean and 90th percentile concentrations of 0.000340 and 0.001391 µg/L, respectively. The mean and 90th percentile concentrations for total PCBs in suspended solids were estimated to be 50 and 107 µg/kg dw TSS, respectively (Table 7-10). The highest concentrations of total PCBs were detected during two of the three sampled storm events (May 2007 and December 2007); at lower flow rates there does not appear to be any correlation between total PCB concentration and flow rate (Figure 7-8), although the relationship was not evaluated statistically. Total PCB concentrations during the third storm event (June 2008) were lower than during the other two storm events and some of the other sampling events. The other sampling events for PCBs and other analytes were conducted during routine monthly monitoring and may or may not correspond to rainfall events. Also, the duration and extent of the wet weather events were not evaluated when making this observation.

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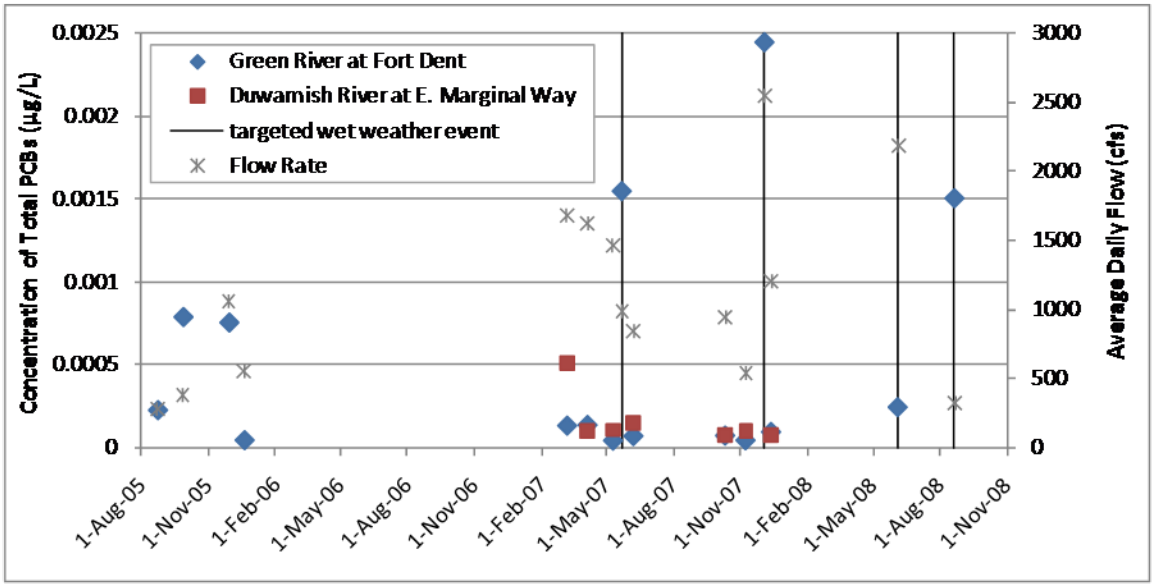
[124] TSS-normalized total PCB concentrations are conservative estimates of particulate-associated PCB concentrations because unfiltered samples likely include dissolved PCBs as well as PCBs associated with solids.



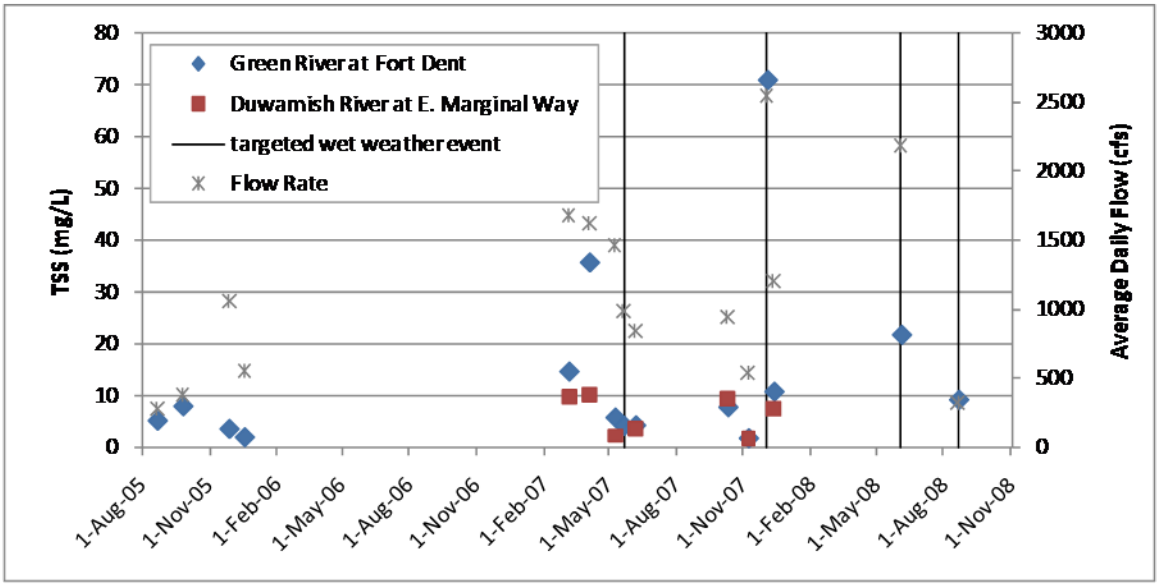


Figure 7-8. Total PCB and TSS concentrations in surface water at upstream Green /Duwamish River locations

For comparative purposes, total PCB concentrations in surface water from the LDW were also normalized using TSS concentrations to evaluate the PCB concentrations associated with total solids (Table 7-11). The highest flow rate was associated with the lowest TSS-normalized total PCB concentration at each location and depth, with a similar pattern for total PCB concentrations. The lowest flow rate was associated with the highest TSS-normalized total PCB concentration at two of the four locations and depths (RM 0.0 at 1 m above the bottom and RM 3.3 at 1 m below the surface).

Table 7-11. Total PCB and TSS concentrations in LDW surface water samples and estimated total PCB concentrations associated with suspended solids

		Surface Water				Suspended Solids (estimated)	
		Total PCB (µg/L)[b]		TSS (mg/L)		Total PCBs (µg/kg TSS)	
Date	Daily Average Flow (cfs)[a]	LDW at RM 0.0	LDW at RM 3.3	LDW at RM 0.0	LDW at RM 3.3	LDW at RM 0.0	LDW at RM 3.3
August 22, 2005							
1 m below the surface	277	0.001796	0.001592 J	4.80	3.40	370	470
1 m above the bottom		0.001814	0.003211	3.1	11.1	590	290
September 26, 2005							
1 m below the surface	378	0.001024	0.001452 J	6.00	5.00	170	290
1 m above the bottom		na	0.001883 J	3.7	5.8	na	320
November 28, 2005							
1 m below the surface	1,060	0.000591	0.000398	4.20	4.30	140	93
1 m above the bottom		0.00025	0.000132	2.0	4.20	130	31
December 19, 2005							
1 m below the surface	550	0.001947 J	0.001122	5.05	4.34	390	260
1 m above the bottom		0.000599	0.001341	2.9	3.7	210	360

[a] Flow data are from the USGS station 12113000 at Auburn, Washington.

[b] Total PCB concentrations were calculated as the sum of detected PCB congeners, which is consistent with the totaling method used for total PCBs based on Aroclors.

cfs – cubic feet per second
dw – dry weight
LDW – Lower Duwamish Waterway
na – not available
PCB – polychlorinated biphenyl
RM – river mile
TSS – total suspended solids

7.2.4.2 Arsenic

King County collected surface water samples from the Green River at Fort Dent from 2001 to 2006 for arsenic analysis (Map 7-7). Only samples with paired total and dissolved arsenic and TSS data, and with total arsenic concentrations greater than dissolved arsenic concentrations, were used to estimate arsenic concentrations associated with suspended solids. Particulate arsenic concentrations in water were calculated by subtracting (on a per-sample basis) dissolved arsenic from total arsenic concentrations in water. If, in a given sample, total arsenic was detected but dissolved arsenic was not, one half the method detection limit (MDL) for the dissolved arsenic was subtracted from the detected total arsenic concentration. The resulting particulate arsenic value was then normalized to the sample-specific TSS concentrations to estimate the arsenic concentrations in suspended solids.

Summary statistics were calculated for total and dissolved arsenic concentrations in surface water and for estimated arsenic concentrations associated with suspended solids. The mean and 90th percentile concentrations of total arsenic were 0.87 and 1.2 µg/L, respectively; the mean and 90th percentile concentrations of dissolved arsenic were 0.52 and 0.71 µg/L, respectively (Table 7-12). The mean and 90th percentile concentrations of arsenic normalized to suspended solids were 37 and 73 mg/kg dw TSS, respectively (Table 7-12).[125] The relatively high arsenic concentrations (compared with other arsenic concentrations summarized in Table 7-7) associated with suspended solids in water samples may be a reflection of high arsenic concentrations on very fine particulates distributed throughout the drainage basin by atmospheric deposition from the Asarco smelter in Ruston, Washington. The highest concentration of total arsenic was detected during the highest flow sampled, but there does not appear to be any significant relationship between arsenic concentration and flow in the rest of the dataset (Figures 7-9 through 7-11).

Table 7-12. Arsenic and TSS concentrations in upstream surface water samples and estimated arsenic concentrations associated with suspended solids

		SURFACE WATER – GREEN RIVER AT FORT DENT				SUSPENDED SOLIDS (estimated)
DATA TYPE	DAILY AVERAGE FLOW (cfs)[a]	TOTAL ARSENIC (µg/L)	DISSOLVED ARSENIC (µg/L)	PARTICULATE ARSENIC (total arsenic minus dissolved arsenic) (µg/L)[b]	TSS (mg/L)	ARSENIC (mg/kg TSS)[c]
Individual Events						
April 30, 2001 – storm[d]	2,250	0.76	0.5 U	0.51	11.7	**44**
June 11, 2001 – storm[d]	1,290	0.9	0.63	0.27	9.2	29
November 14, 2001 – storm[d]	3,800	1.1	0.62	0.48	27	18
January 24, 2002 – storm[d]	2,220	0.576	0.44	0.14	7.3	19
February 13, 2002	1,370	0.69	0.5 U	0.44	3.82	**120**
		0.68	0.5 U	0.43	3.75	**110**
		0.65	0.5 U	0.40	4.62	**87**
February 21, 2002 – storm[d]	2,030	0.72	0.71	0.01	20.1	0.50
		0.77	0.67	0.10	16.6	6.0
February 22, 2002 – storm[d]	3,700	1.1	0.81	0.29	53	5.5
		1.3	0.79	0.51	53.6	9.5
		0.84	0.85	na[b]	39.3	na
		0.5 U	0.79	na[b]	25.2	na
March 11, 2002 – storm[d]	1,200	0.76	0.54	0.22	8.4	26
March 25, 2002	1,550	0.75	0.53	0.22	7.11	31
		0.74	0.54	0.20	5.8	34
March 26, 2002	1,520	0.75	0.53	0.22	12.4	18

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[125] Summary statistics for TSS-normalized arsenic concentrations were calculated using ProUCL with non-detects set at full reporting limit if dissolved arsenic was not detected. Using MTCAStat, the 90th percentile concentration of TSS-normalized arsenic would have been 56 mg/kg dw TSS if the full MDL had been used or 88 mg/kg dw TSS if zero had been used for non-detected dissolved arsenic concentrations.

DATA TYPE	DAILY AVERAGE FLOW (cfs)[a]	SURFACE WATER – GREEN RIVER AT FORT DENT				SUSPENDED SOLIDS (estimated)
		TOTAL ARSENIC (µg/L)	DISSOLVED ARSENIC (µg/L)	PARTICULATE ARSENIC (total arsenic minus dissolved arsenic) (µg/L)[b]	TSS (mg/L)	ARSENIC (mg/kg TSS)[c]
April 24, 2002	2,030	0.64	0.5 U	0.39	8.79	**44**
		0.63	0.5 U	0.38	10.8	**35**
April 25, 2002	1,880	0.59	0.5 U	0.34	9.89	**34**
June 12, 2002	1,700	0.5 U	0.5 U	na[b]	7.47	na
		0.5 U	0.5 U	na[b]	6.49	na
		0.57	0.5 U	0.32	8.02	**40**
June 28, 2002 – storm[d]	952	0.73	0.55	0.18	6.2	29
		0.86	0.54	0.32	11	29
		0.65	0.51	0.14	5.1	27
June 29, 2002 – storm[d]	1,210	0.68	0.56	0.12	5.8	21
		0.72	0.52	0.20	11.5	17
		0.72	0.51	0.21	10	21
		0.75	0.55	0.20	8.3	24
		0.75	0.54	0.21	9.4	22
		0.71	0.53	0.18	6.2	29
June 30, 2002 – storm[d]	1,390	0.74	0.56	0.18	8.9	20
August 6, 2002	294	0.933	0.741	0.19	3.4	56
October 23, 2002	244	0.98	0.78	0.20	2.27	88
November 6, 2002 – storm[d]	209	0.91	0.62	0.29	4.9	59
		0.91	0.65	0.26	4.24	61
November 7, 2002 – storm[d]	207	0.85	0.65	0.20	2.5	80
		0.79	0.6	0.19	2.6	73
November 12, 2002 – storm[d]	264	0.98	0.66	0.32	3	110
December 3, 2002	197	0.71	0.55	0.16	1.2	130
December 4, 2002	195	0.76	0.55	0.21	2.3	91
December 11, 2002 – storm[d]	213	0.72	0.54	0.18	4.29	41
		0.65	0.51	0.14	4.3	33
December 12, 2002 – storm[d]	371	1.1	0.57	0.53	12.4	43
		0.88	0.55	0.33	11.3	29
		0.67	0.61	0.060	5.05	12
		0.71	0.52	0.19	4.79	40
		0.76	0.53	0.23	6.38	36
		0.67	0.54	0.13	5.06	26
December 13, 2002 – storm[d]	816	1.1	0.58	0.52	9.9	53
January 3, 2003 – storm[d]	1,220	1.2	0.64	0.56	21.6	26

		Surface Water – Green River at Fort Dent				Suspended Solids (estimated)
Data Type	Daily Average Flow (cfs)[a]	Total Arsenic (µg/L)	Dissolved Arsenic (µg/L)	Particulate Arsenic (total arsenic minus dissolved arsenic) (µg/L)[b]	TSS (mg/L)	Arsenic (mg/kg TSS)[c]
January 4, 2003 – storm[d]	1,150	1	0.62	0.38	14.9	26
		0.98	0.67	0.31	13.1	24
		1.2	0.53	0.67	18.6	36
		1.1	0.63	0.47	16.1	29
		1	0.63	0.37	19.6	19
		1.2	0.59	0.61	21.2	29
January 5, 2003 – storm[d]	1,420	1	0.67	0.33	15.8	21
		1	0.69	0.31	17.7	18
		1	0.64	0.36	16.4	22
January 21, 2003 – storm[d]	646	0.71	0.53	0.18	4.5	40
		0.71	0.51	0.20	4.4	45
		0.78	0.5 U	0.53	6.79	**78**
January 22, 2003 – storm[d]	1,040	0.89	0.58	0.31	11.8	26
		0.8	0.56	0.24	9.85	24
		0.78	0.5	0.28	8.98	31
		0.76	0.5	0.26	7.88	33
		0.73	0.5 U	0.48	6.3	**76**
January 23, 2003 – storm[d]	1,320	0.95	0.63	0.32	13.8	23
January 27, 2003 – storm[d]	5,210	1.6	0.5 U	1.4	52.4	**26**
February 12, 2003	1,220	0.65	0.5 U	0.40	7.94	**50**
		0.71	0.5 U	0.46	15.6	**29**
February 13, 2003	1,110	0.63	0.5 U	0.38	8.37	**45**
March 8, 2003 – storm[d]	1,130	0.68	0.53	0.15	6.9	22
		0.69	0.57	0.12	6.18	19
		0.68	0.52	0.16	5.24	31
March 9, 2003 – storm[d]	1,340	0.71	0.53	0.18	10.4	17
		0.64	0.5 U	0.39	6.8	**57**
		0.66	0.55	0.11	5.39	20
		0.71	0.54	0.17	10.4	16
		0.73	0.52	0.21	10.4	20
		0.73	0.51	0.22	10	22
March 10, 2003 – storm[d]	2,040	0.77	0.52	0.25	14.3	17
April 28, 2003	1,540	0.57	0.5 U	0.32	8.12	**39**
		0.53	0.5 U	0.28	17	**16**
April 29, 2003	1,400	0.57	0.5 U	0.32	7.22	**44**
June 8, 2003	662	0.91	0.62	0.29	10.4	28
		0.93	0.67	0.26	7.84	33
June 9, 2003	655	0.85	0.7	0.15	12	13

DATA TYPE	DAILY AVERAGE FLOW (cfs)[a]	SURFACE WATER – GREEN RIVER AT FORT DENT				SUSPENDED SOLIDS (estimated)
		TOTAL ARSENIC (µg/L)	DISSOLVED ARSENIC (µg/L)	PARTICULATE ARSENIC (total arsenic minus dissolved arsenic) (µg/L)[b]	TSS (mg/L)	ARSENIC (mg/kg TSS)[c]
August 26, 2003	234	1.1	0.81	0.29	6.63	44
		1.1	0.8	0.30	6.4	47
August 27, 2003	230	1.2	0.84	0.36	6.1	59
October 16, 2003 – storm[d]	546	1.1	0.61	0.49	12.9	38
October 17, 2003 – storm[d]	1,050	1.5	0.58	0.92	28.2	33
November 17, 2003 – storm[d]	1,200	0.85	0.5 U	0.60	15.8	**38**
November 18, 2003 – storm[d]	3,650	1.6	0.5 U	1.4	67	**20**
May 26, 2004 – storm[d]	957	0.82	0.53	0.29	7.6	38
September 13, 2004 – storm[d]	682	1.0	0.69	0.31	11.9	26
January 18, 2005 – storm[d]	6,050	1.6	0.61	0.99	63.2	16
October 31, 2005 – storm[d]	1,070	0.72	0.57	0.15	6.6	23
January 9, 2006 – storm[d]	4,540	0.99	0.5 U	0.74	34.4	**22**
September 19, 2006 – storm[d]	297	1.0	0.69	0.31	9.8	32
November 7, 2006 – storm[d]	10,200	4.16	0.5 U	3.9	312	**13**
Summary Statistics[f]						
Number of samples	55	104	104	100	104	100
Detection frequency	na	101/104	80/104	na[e]	104/104	100/100
Mean	1,580	0.87	0.52	0.37	15	37[g]
Minimum	195	0.25	0.25	0.01	1.2	0.50[g]
Maximum	10,200	4.2	0.85	3.9	312	130[g]
RL or range of RLs	na	0.5	0.5	na	na	na
25th Percentile	646	0.70	0.48	0.19	5.5	21[g]
50th Percentile	1,200	0.76	0.55	0.29	9.6	29[g]
75th Percentile	1,700	0.99	0.63	0.39	17	43[g]
90th Percentile	3,740	1.2	0.71	0.56	28	73[g]

[a] Flow data are from the USGS station 12113000 at Auburn, Washington.

[b] If, in a given sample, total arsenic was detected but dissolved arsenic was not, one-half the MDL for the dissolved arsenic was subtracted from the total arsenic concentration to estimate the particulate arsenic concentration. Particulate arsenic concentrations could not be calculated for four samples because: 1) the dissolved concentration was greater than the total concentration (two samples on February 22, 2002), or 2) neither total nor dissolved arsenic was detected (two samples on June 12, 2002).

[c] Arsenic concentrations associated with suspended solids (in mg/kg TSS) were calculated as follows:

$$\frac{\mathrm{As}_{\mathrm{Total}}\left(\frac{\mu g}{L}\right) - \mathrm{As}_{\mathrm{Dissolved}}\left(\frac{\mu g}{L}\right)}{\mathrm{TSS}\left(\frac{mg}{L}\right)}\left(1{,}000{,}000\,\frac{mg}{kg}\right)\left(\frac{mg}{1000\,\mu g}\right) = \mathrm{As}\left(\frac{mg}{kg}\mathrm{TSS}\right)$$

[d] A storm was a targeted event with at least 0.25-in. rainfall over 24 hours and a preceding minimum dry period of 24 hours. The other sampling events were conducted during routine monthly monitoring and may or may not correspond to storm events.

[e] Detection frequency is not applicable for particulate arsenic because it is a calculated concentration using one-half the MDL if dissolved arsenic was not detected (see Footnote b).

[f] To evaluate the influence of non-detected dissolved arsenic concentrations in the calculation of the suspended solid arsenic concentrations, summary statistics were also calculated using MTCAStat assuming non-detects were equal to 0, non-detects were equal to the full MDL, and non-detects were excluded from the dataset, as shown below:

	Arsenic concentration (mg/kg TSS)			Percentiles for arsenic concentrations (mg/kg TSS)			
Treatment of NDs	Minimum	Maximum	Mean	25th	50th	75th	90th
ND = 1/2 MDL	0.50	130	37	21	29	43	76
ND = 0	0.50	180	43	22	30	56	88
ND = full MDL	0.50	130	30	18	26	36	56
Excluding ND data	0.50	130	33	20	28	38	59

The maximum concentration of 180 mg/kg TSS was calculated using data from the first sample collected on February 13, 2002, and zero for non-detects.

[g] The mean and percentile concentrations were calculated using statistical software ProUCL 4.00.04 (EPA 2007e), with non-detects set at full reporting limit.

cfs – cubic feet per second
dw – dry weight
MDL – method detection limit
na – not applicable
ND – non-detect value
TSS – total suspended solids
U – not detected at reporting limit shown

Bold identifies TSS-normalized arsenic concentrations that were calculated using non-detected dissolved arsenic concentrations assuming one-half the MDL for those values.

For comparative purposes, Table 7-13 presents TSS-normalized arsenic concentrations in surface water from the LDW. These concentrations were calculated by subtracting (on a per-sample basis) dissolved arsenic from total arsenic concentrations in surface water. The resulting particulate arsenic concentration was then divided by the sample-specific TSS concentration. TSS-normalized arsenic concentrations ranged from 0.5 to 340 mg/kg TSS in the LDW, with means for the three locations ranging from 12 to 25 mg/kg TSS.

Table 7-13. Summary of estimated arsenic concentrations associated with suspended solids in LDW surface water

	Detection Frequency[b]		Arsenic Concentration (mg/kg TSS)			TSS Concentration (mg/L)		
Location ID[a]	Ratio	%	Minimum	Maximum	Mean	Minimum	Maximum	Mean
BRN (RM 1.1)	60/60	100	0.5	340	25	2.6	69	11
SWM (RM 1.9)	59/59	100	2.3	100	20	5.5	31.3	15
NFK (RM 4.9)	22/22	100	3.8	29	12	2.6	48.7	11

[a] BRN, SWM, and NFK are located in the LDW near the Brandon, SW Michigan, and Norfolk CSOs, respectively.

[b] TSS-normalized arsenic concentrations were calculated for surface water samples that were analyzed for both dissolved and total arsenic, In those samples, both dissolved and total arsenic were always detected.

CSO – combined sewer overflow
dw – dry weight
ID – identification
LDW – Lower Duwamish Waterway
RM – river mile
TSS – total suspended solids

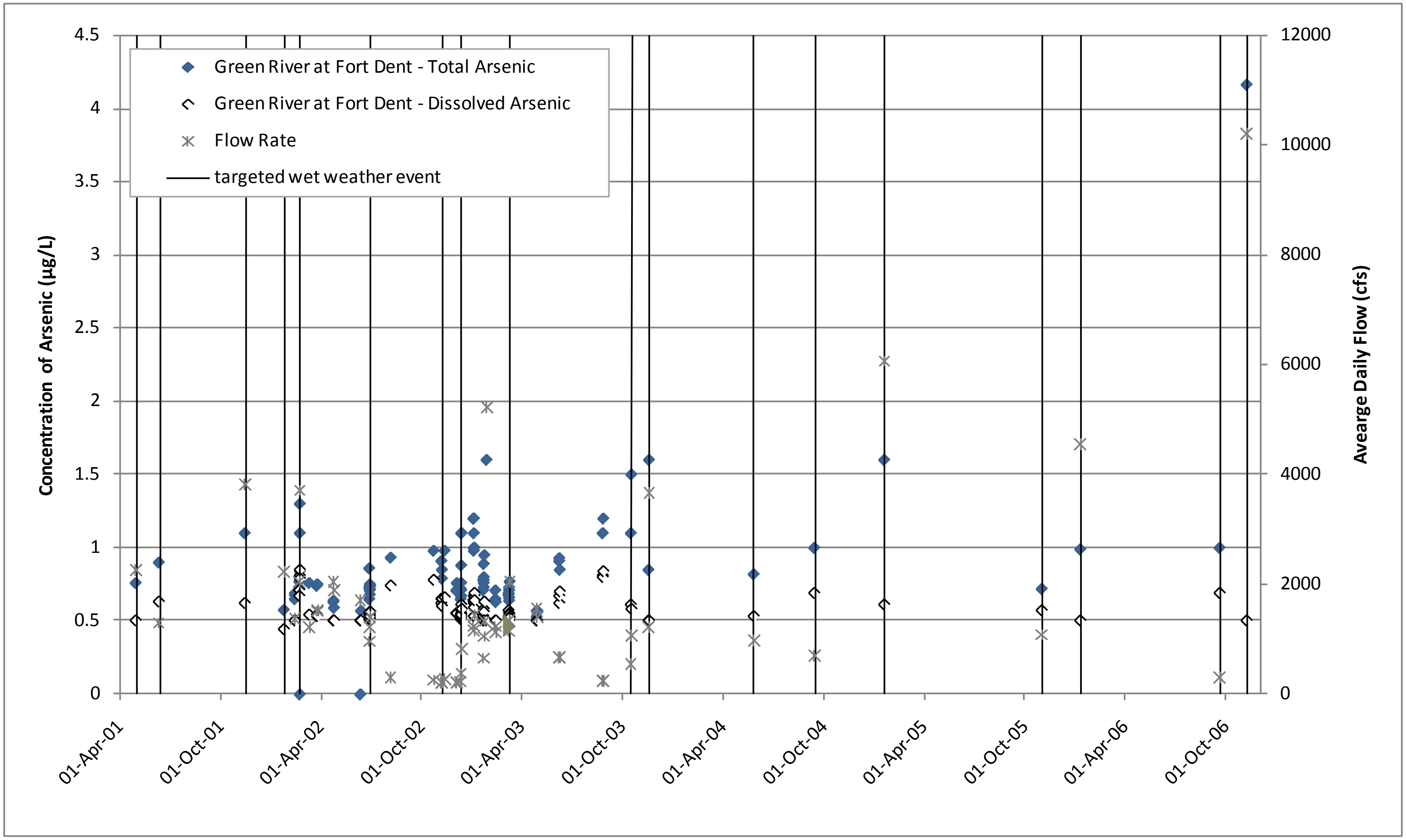

Figure 7-9. Arsenic concentrations in surface water at the upstream Green River at Fort Dent location

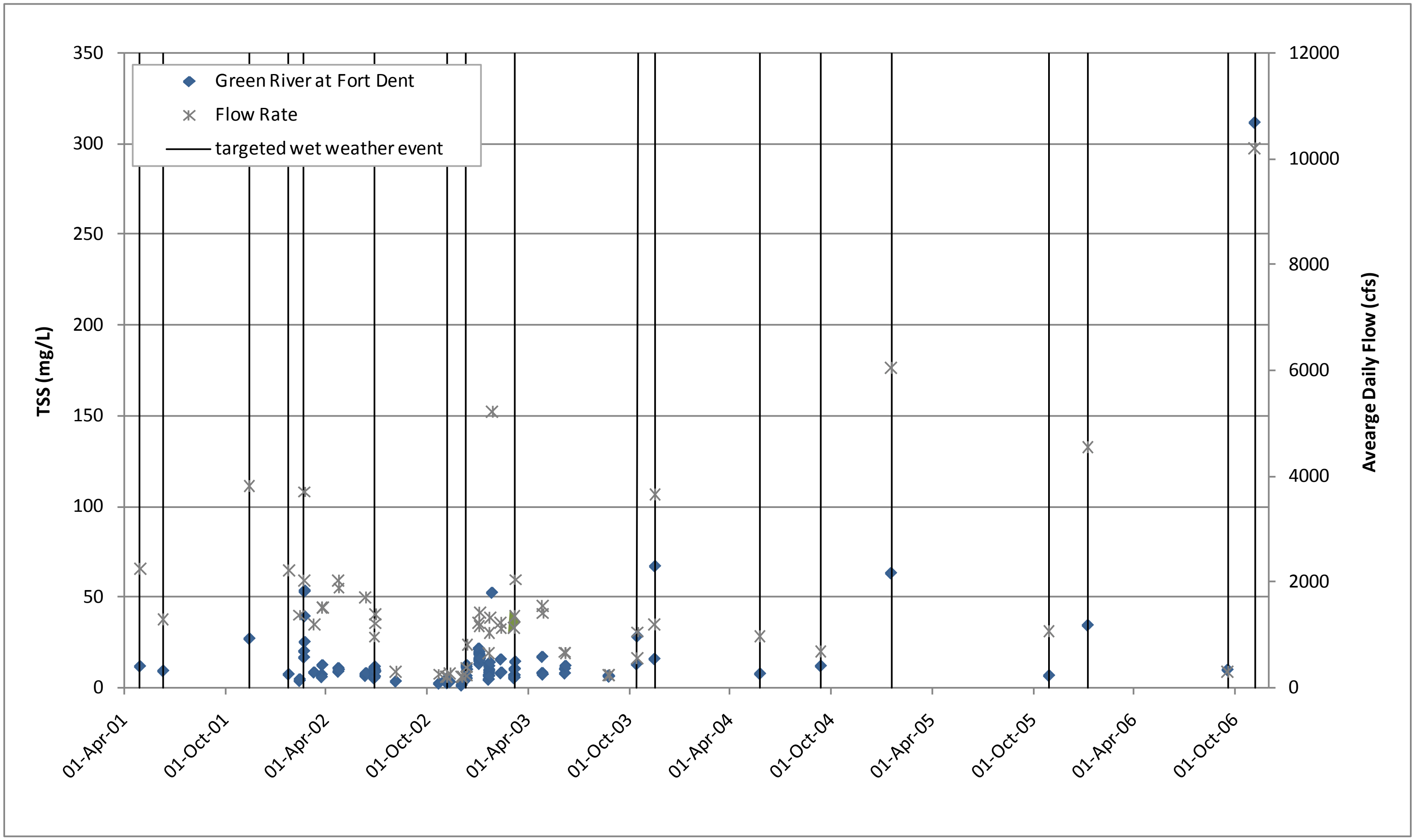

Figure 7-10. TSS concentrations in surface water at the upstream Green River at Fort Dent location

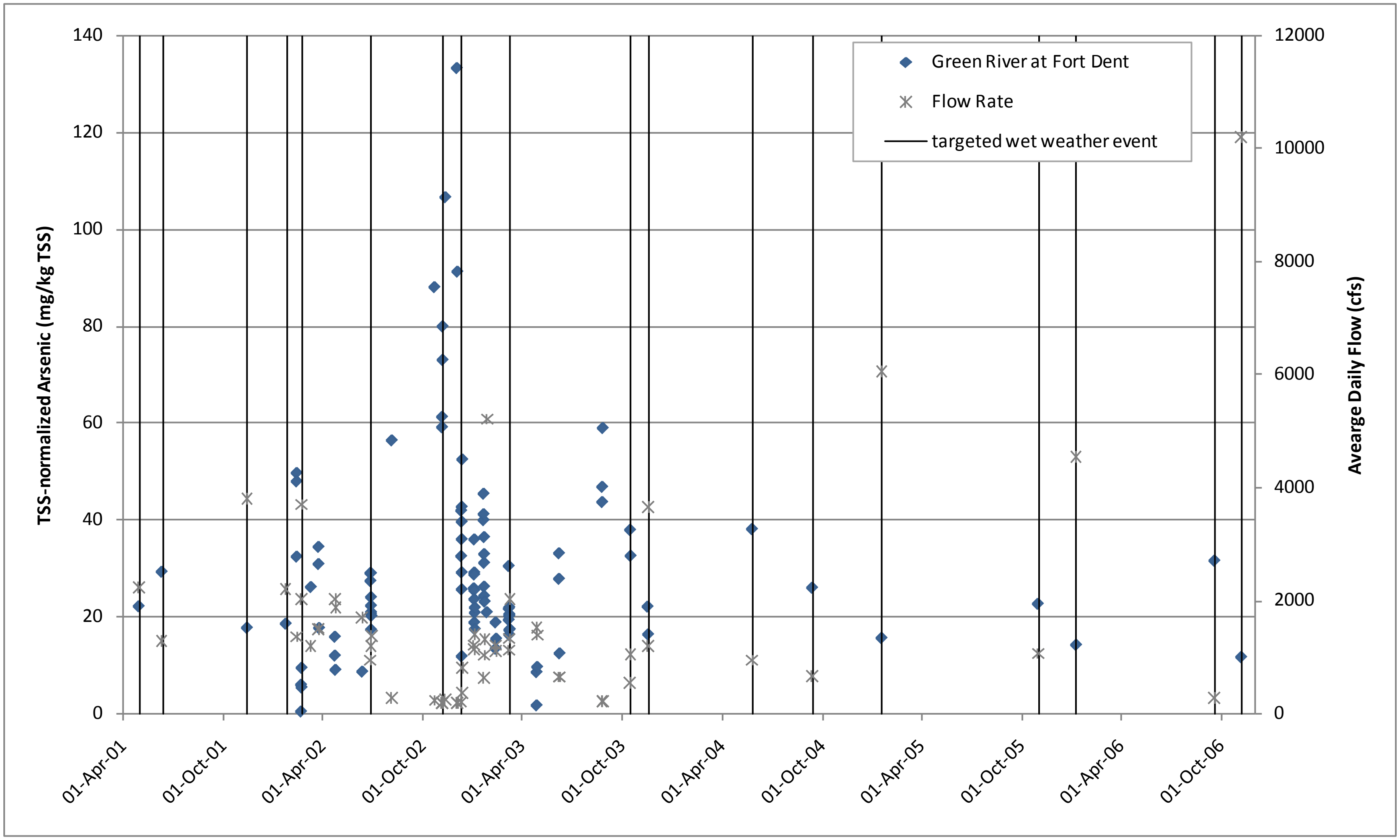

Figure 7-11. TSS-normalized arsenic concentrations in surface water at the upstream Green River at Fort Dent location

7.2.4.3 cPAHs

From 2001 through 2003, King County collected unfiltered surface water samples for PAH analysis from the Green River at Fort Dent (Map 7-7). PAHs were not detected in these samples (Herrera 2004b, 2005). In 2008, King County collected unfiltered surface water samples from the Green River at Fort Dent and from the Duwamish River at the E Marginal Way S Bridge (at S 155th Street, RM 6.3) monthly from February through August 2008. In addition, samples were also collected during two storm events (June and August) from the Green River at Fort Dent (Table 7-14; Map 7-7). These samples were analyzed using a method with lower reporting limits for PAHs (EPA Method 8270D-SIM); TSS was also analyzed in these samples.

Table 7-14. cPAH and TSS concentrations in upstream surface water samples and estimated cPAH concentrations associated with suspended solids

Data Type	Daily Average Flow (cfs)[a]	Surface Water				Suspended Solids (estimated)	
		cPAH (µg/L)[b]		TSS (mg/L)		cPAH (µg/kg TSS)[c]	
		Duwamish River at E Marginal Way S	Green River at Fort Dent	Duwamish River at E Marginal Way S	Green River at Fort Dent	Duwamish River at E Marginal Way S	Green River at Fort Dent
Individual Events							
February 6, 2008	837	0.0016	0.0011[d]	3.9	5.2[d]	408	220[e]
March 5, 2008	1,810	0.00045 U[b, d]	0.00045 U[b]	8.5[d]	9.8	nc[e]	nc
March 17, 2008	1,850	0.00049[d]	0.00051	7.7[d]	9.6	64[e]	53
April 9, 2008	1,180	0.0010	0.0005[d]	4.0	4.2[d]	239	117[e]
May 7, 2008	2,470	0.00104[d]	0.0011	19.3[d]	20.8	53.8[e]	53.2
June 3, 2008 – storm[f]	2,190	no data	0.0027[d]	no data	21.8[d]	no data	122[e]
June 12, 2008	3,570	0.00045 U[b]	0.00046[d]	20.4	15.9[d]	nc	29[e]
July 9, 2008	1,330	0.00045 U[b,d]	0.00045 U[b]	1.9[d]	6.1	nc[e]	nc
August 6, 2008	341	0.00048	0.00045 U[b,d]	1.4	2.5[d]	346	nc`
August 20, 2008 – storm[f]	321	no data	0.0035[d]	no data	9.2[d]	no data	384[e]
Summary Statistics							
Number of samples	na	18		18		18	
Detection frequency	na	12/18		18/18		12/18	
Mean detect	1,590	0.0012		9.6		151[h]	
Minimum detect	321	0.00046		1.4		29	
Maximum detect	3,570	0.0035		21.8		408	
25th Percentile	713	0.00049[g]		4.0[g]		53[h]	
50th Percentile	1,520	0.0010[g]		8.1[g]		74[h]	
75th Percentile	2,260	0.0015[g]		17[g]		229[h]	
90th Percentile	3,460	0.0032[g]		21[g]		354[h]	

[a] Flow data are from the USGS Station 12113000 at Auburn, Washington.

[b] cPAH was calculated as the sum of each individual cPAH concentration multiplied by the corresponding PEF, as discussed in more detail in Appendix E, Section E.3. When the individual cPAH component concentration was reported as

non-detected, then the PEF was multiplied by half the MDL. A U-qualifier indicates that no individual cPAH components were detected.

[c] TSS-normalized cPAHs were calculated by dividing the cPAH concentration (in µg/L) by the TSS concentration (in kg/L). TSS-normalized cPAHs were not calculated for samples with only U-qualified cPAH results.

[d] Field replicates were collected for these samples; the average cPAH and TSS concentrations shown represent averages of the original and field replicates.

[e] Field replicates were collected for these samples. The average TSS-normalized cPAH concentrations were first calculated separately for the original and the field replicate, and then the two concentrations were averaged.

[f] A storm was a targeted event with at least 0.25-in. rainfall over 24 hours, with a minimum 24-hour antecedent dry. The other sampling events were conducted during routine monthly monitoring and may or may not correspond to storm events.

[g] Percentiles were calculated using methods presented in the National Institute of Standards and Technology Handbook of Statistical Methods ((NIST and SEMATECH 2006).

[h] The mean and percentile concentrations were calculated using statistical software ProUCL 4.00.04 (EPA 2007e), with non-detects set at full reporting limit.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
cfs – cubic feet per second
MDL – method detection limit
na – not applicable
nc –not calculated
PEF – potency equivalency factor
TSS – total suspended solids
U – not detected at given concentration

Summary statistics were calculated for cPAH in surface water and estimated for cPAH associated with suspended solids. Detected cPAH concentrations in surface water samples ranged from 0.0005 to 0.0035 µg/L. The mean and 90th percentile cPAH concentrations normalized to TSS were 151 and 354 µg/kg dw TSS, respectively (Table 7-14).[126] There does not appear to be any significant relationship between cPAH concentration and flow in the dataset (Figure 7-12). cPAH concentrations associated with suspended solids could not be calculated for surface water from the LDW because the LDW cPAH surface water data were calculated from SPMD samples (without corresponding TSS data).

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[126] When the six surface water samples with no detected cPAHs were excluded from the calculations, the mean and 90th percentile cPAH concentrations normalized to TSS were 159 and 388 µg/kg dw TSS using the National Institute of Standards and Technology Handbook of Statistical Methods (NIST and SEMATECH 2006).

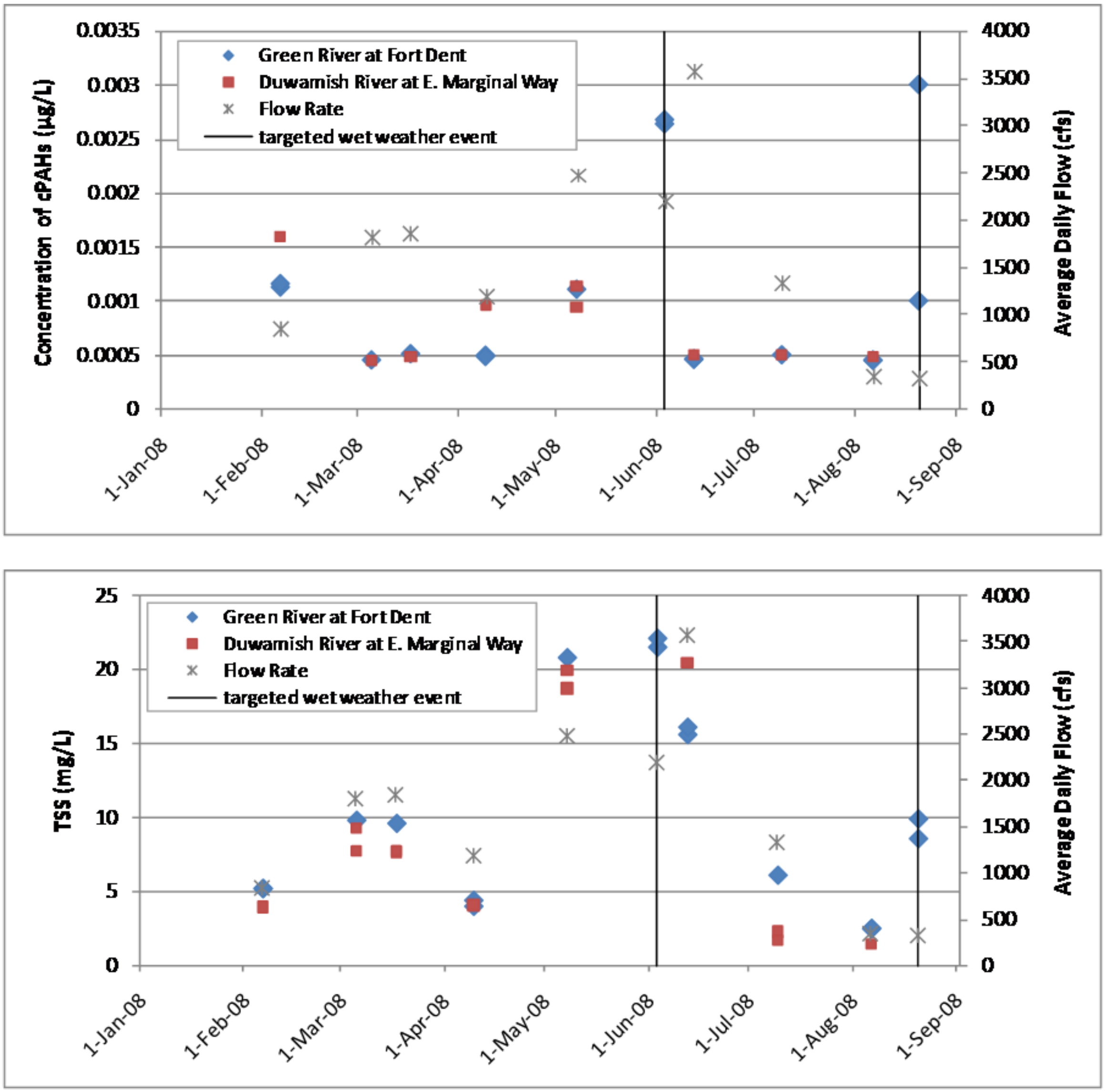

Figure 7-12. Carcinogenic PAH and TSS concentrations in surface water at upstream Green River/Duwamish locations

7.3 URBAN BACKGROUND SEDIMENT DATA

To provide additional context for the sediment concentrations presented in Sections 7.1 and 7.2, this section presents concentrations of total PCBs, arsenic, cPAHs, and dioxins and furans from locations outside the Green/Duwamish watershed. These data are being presented only for informational purposes (i.e., they are not being used to define background for deriving sediment cleanup values).

7.3.1 PCBs, arsenic, and cPAHs

Surface sediment data for total PCBs, arsenic, and cPAHs within the following major urban bays and lakes were extracted from Ecology's EIM system: Elliott Bay, Bellingham Bay, Commencement Bay, Lake Washington, and Lake Sammamish. These data include the results from many individual studies. As noted in the introduction to Section 7, detailed evaluations of dataset representativeness were not performed.

Prior to the calculation of statistics, these data were screened to exclude the following types of samples:

- Samples collected as part of a CERCLA or MTCA cleanup study prior to any sediment remediation (post-remediation monitoring data from any such sites were retained)
- Samples collected as part of routine onsite monitoring of the DMMP open-water dredged material disposal sites

Although data from the open-water dredged material disposal sites may be representative of regional sediment quality, these data were excluded because of the difficulty in attributing the sediment characteristics of the dredged material to the original location from which the sediments were dredged.

Elliott Bay locations were divided into an inner Elliott Bay dataset and an outer Elliott Bay dataset. The inner and outer datasets were defined by drawing a north-south line from the Duwamish head to Pier 91/92. Three locations near the middle and east of the line were included in the outer Elliott Bay dataset because the water depths were greater than 140 m. The Elliott Bay datasets include data collected from 1991 to 1998 but do not include post-cap monitoring samples from Pier 52/53 and Denny Way. Inner Elliott Bay data are included in Tables 7-15 through 7-17 for informational purposes but are excluded from the summary ranges discussed in this section and in Section 7.5 because inner Elliott Bay receives discharge from the LDW and may be influenced by other known point sources of contamination. Additional screening steps were applied with the objective of generating a single result per chemical for each location, which involved averaging field duplicates, field replicates, and laboratory replicates, as described in Appendix E. In addition, for locations monitored repeatedly over time, only the most recent result was included. Although additional screening steps were discussed and evaluated with EPA and Ecology (RETEC 2007a, b, c), only those steps discussed above were ultimately conducted to derive the datasets presented in this section.

The summary statistics for each urban bay or lake were evaluated individually (Tables 7-15, 7-16, and 7-17). The available data for each water body from studies not specifically designed as background evaluations may not be representative of overall conditions (or that portion that might be deemed to reflect background) in that water body. Differences in available datasets from one water body to another may also affect the comparability of results across different areas. In addition, aside from the initial screening discussed above, these data were not thoroughly screened to ensure that all data that might be associated with hazardous waste sites or other point sources of contamination were removed. Therefore, the data summarized in this section are presented only for informational purposes.

Table 7-15. Total PCB concentrations in surface sediment samples collected in urban bays and lakes, excluding cleanup and disposal sites

AREA	DETECTION FREQUENCY	PCB CONCENTRATION (µg/kg dw)							
		MINIMUM DETECT	MAXIMUM DETECT	MEAN	95% UCL	50TH PERCENTILE	90TH PERCENTILE	MINIMUM NON-DETECT	MAXIMUM NON-DETECT
Inner Elliott Bay	28/37	33	800	190	255	99	576	18	59
Outer Elliott Bay	7/28	8.1	138	38	53	17	82	10	844
Bellingham Bay	6/61	8.0	425	76	164	25	114	15	850
Commencement Bay	49/71	4.0	1,104	61	127	21	64	12	350
Lake Sammamish	25/25	16	88	40	49	34	73	na	na
Lake Washington	1/17	26	26	87	137	47	217	9.5	285

Note: Mean and percentile values were calculated using both detected and non-detected concentrations, where non-detects were set equal to one-half the non-detect concentration. The 95% UCL values were calculated in ProUCL v. 4.0, using both detect and non-detect concentrations.

dw – dry weight

na – not applicable

PCB – polychlorinated biphenyl

UCL – upper confidence limit on the mean

Table 7-16. Arsenic concentrations in surface sediment samples collected in urban bays and lakes, excluding cleanup and disposal sites

		Arsenic Concentration (mg/kg dw)							
Area	Detection Frequency	Minimum Detect	Maximum Detect	Mean	95% UCL	50th Percentile	90th Percentile	Minimum RL	Maximum RL
Inner Elliott Bay	25/34	4.7	27	8.6	10.4	7.4	16	6.0	11
Outer Elliott Bay	19/31	2.4	14	5.1	6.4	4.1	9.8	2.9	18
Bellingham Bay	160/162	1.5	19	9.2	9.6	9.2	13	3.3	7.5
Commencement Bay	131/133	1.4	45	9.6	12	8.7	17	13	13
Lake Washington	25/29	2.0	27	7.2	8.9	6.3	13	4.0	20
Lake Sammamish	29/29	1.8	72	15	59	8.7	38	na	na

Note: Mean and percentile values were calculated using both detected and non-detected concentrations, where non-detects were set equal to one-half the RL. The 95% UCL values were calculated in ProUCL v. 4.0, using both detect and non-detect concentrations.

dw – dry weight

na – not applicable

RL – reporting limit

UCL – upper confidence limit on the mean

Table 7-17. cPAH concentrations in surface sediment samples collected in urban bays and lakes, excluding cleanup and disposal sites

AREA	DETECTION FREQUENCY	cPAH CONCENTRATION (µg/kg dw)							
		MINIMUM DETECT	MAXIMUM DETECT	MEAN	95% UCL	50TH PERCENTILE	90TH PERCENTILE	MINIMUM NON-DETECT	MAXIMUM NON-DETECT
Inner Elliott Bay	64/66	14	4,780	583	1,080	269	1,410	34	36
Outer Elliott Bay	15/21	22	327	116	152	79	292	20	39
Bellingham Bay	53/64	5.8	593	76	108	32	185	15	26
Commencement Bay	45/45	8.8	1,700	223	345	115	527	na	na
Lake Washington	30/33	43	5,290	374	635	216	904	49	855
Lake Sammamish	11/20	57	1,870	234	407	92	574	41	453

Note: Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3. Mean and percentile values were calculated using both detected and non-detected concentrations, where non-detects were set equal to one-half the non-detect concentration. The 95% UCL values were calculated in ProUCL v. 4.0, using both detect and non-detect concentrations.

cPAH – carcinogenic polycyclic aromatic hydrocarbons

dw – dry weight

na – not applicable

PEF – potency equivalency factor

UCL – upper confidence limit on the mean

Excluding the data from inner Elliott Bay, the mean total PCB concentrations for sediments in urban bays and lakes ranged from 38 to 87 µg/kg dw. The mean concentrations from these water bodies were within or higher than the range of mean total PCB concentrations from the three datasets evaluated in Section 7.2, which ranged from 17 to 51 µg/kg dw (Table 7-7). The 90th percentiles of the total PCB data in urban bays and lakes ranged from 64 to 217 µg/kg dw, with the exclusion of data from inner Elliott Bay (Table 7-15), compared with 46 to 120 µg/kg dw in the three datasets in Section 7.2.

The mean arsenic concentrations in urban bays and lakes ranged from 5.1 to 15 mg/kg dw. These means are within the range of arsenic means from the three datasets evaluated in Section 7.2 (7 to 37 mg/kg dw) (Table 7-7). The 90th percentiles of the arsenic data in urban bays and lakes ranged from 9.8 to 38 mg/kg dw (Table 7-16) compared with 11 to 76 mg/kg dw in the three datasets in Section 7.2.

Excluding the data from inner Elliott Bay, the mean cPAH concentrations in urban bays and lakes ranged from 76 to 374 µg/kg dw. The mean concentrations from these water bodies were within or higher than the range of cPAH means from the three datasets evaluated in Section 7.2, which ranged from 51 to 125 µg/kg dw (Table 7-7). The 90th percentiles of the cPAH data in urban bays and lakes ranged from 185 to 904 µg/kg dw, with the exclusion of data from inner Elliott Bay (Table 7-17), compared with 209 to 340 µg/kg dw in the three datasets in Section 7.2.

7.3.2 Dioxins and furans

Surface sediment sampling for dioxins and furans in the greater Seattle metropolitan area was conducted as part of the RI sampling event in 2005 (Windward 2005d). This Seattle-area study was designed to collect sediment samples and provide dioxin and furan TEQ results near storm drains and other areas receiving runoff associated with typical urban sources. Dioxins and furans from these sources are distributed throughout urban areas via atmospheric deposition followed by stormwater runoff. The total number of samples was relatively small; conclusions drawn from these results for urban levels should be considered in this context.

The criteria used to select sampling areas representative of LDW background were as follows: 1) the area must receive drainage from basins with land uses similar to the LDW, 2) the area must not be located near known industrial point sources of dioxins and furans, 3) the area must represent a range of receiving water environments, and 4) the area must represent a range of stormwater discharge frequencies, volumes, and types similar to those in the LDW. The sampling locations that met these four criteria are shown on Map 7-8. In order to assess the potential influence of outfall proximity on dioxin and furan TEQs in sediment, two samples, one between 30 and 50 ft from the outfall and a second between 100 and 120 ft from the outfall, were collected at each of four of these locations: Elliott Bay (Terminal 91), Lake Union (I-5 bridge), Lake Washington (Renton), and the Ship Canal (Salmon Bay).

Individual dioxin and furan congener data were converted to a dioxin and furan TEQ using concentrations of detected congeners, one-half the RLs for congeners that were not detected in a given sample, and mammalian TEFs (Van den Berg et al. 2006). Data were reviewed to determine the appropriate dataset to represent urban conditions expected for areas with stormwater runoff, such as the LDW. The dioxin and furan TEQ results for all individual samples are provided in Table 7-18. TEQs ranged from 5.46 ng/kg dw (Lake Union) to 187 ng/kg dw (in Salmon Bay). Most of the TEQs were less than 30 ng/kg dw. The two samples collected during this study from upstream of the LDW in the Green River, as shown on Map 7-8, are included in the discussion of upstream sediment sampling (Section 7.2.2).

Table 7-18. Dioxin and furan TEQs in surface sediment samples collected from areas near storm drains and from other areas receiving runoff in the greater Seattle metropolitan area

LOCATION	LOCATION ID	DIOXIN AND FURAN TEQ (ng/kg dw)	VALUES INCLUDED IN CALCULATION OF STATISTICS (ng/kg dw)
Elliott Bay (Terminal 91)[a]	EB-SS2a	13.7 J	16.3
	EB-SS2b	18.9 J	
Lake Union (I-5 bridge)[a]	LU-SS9a	5.46 J	15.8
	LU-SS9b	26.1 J	
Lake Washington (Bothell)	LW-SS3	13.2 J[b]	13.2
Lake Washington (Bellevue)	LW-SS4	14.7 J	14.7
Lake Washington (Renton)[a]	LW-SS5a	14.1 J	14.3
	LW-SS5b	14.5 J	
Ship Canal (Salmon Bay)[a]	SC-SS1a	187 J	-
	SC-SS1b	63.1 J	-
Union Bay (Laurelhurst)	UB-SS8	53.4 J	-
Statistics for greater Seattle area locations (excluding samples from the Ship Canal and Union Bay)			**mean – 14.9**
			90th percentile – 16.3
			95% UCL – 16.0

[a] Two samples were collected, one approximately 30 to 50 ft from the outfall and the other approximately 100 to 120 ft from the outfall.

[b] Reported concentration is the average of two field samples.

dw – dry weight
ID – identification
J – estimated concentration
na – not applicable
TEQ – toxic equivalent
UCL – upper confidence limit on the mean

The summary statistics presented in Table 7-18 include two adjustments to the set of individual results. The data for the Ship Canal and Union Bay sampling locations are not included because the TEQs were higher than the national range of 0.12 to 16.3

ng/kg dw (EPA 2000b).[127] Three of the sampled locations had two samples each. The results of the two samples collected at different distances from the outfall at each of these three locations (Elliott Bay, Lake Union, Lake Washington/Renton) were averaged before calculating summary statistics. Averaging was intended to provide a representation of conditions in the general vicinity of the outfalls. The mean dioxin and furan TEQ for this reduced dataset was 14.9 ng/kg dw, higher than the average from EPA's (2000b) national summary (which included both urban and rural locations), but within the range.

7.4 TISSUE DATA FROM PUGET SOUND

This section presents the ranges of total PCB and arsenic concentrations in tissue samples from non-urban and near-urban areas in Puget Sound.

Relatively few tissue data were available for dioxins and furans and cPAHs. Dioxins and furans were identified as a seafood consumption risk driver based on an assumption of unacceptable risk (no LDW tissue data were available); thus, the limited background tissue data available are not presented. For cPAHs, limited background tissue data were available from King County (King County 1999e). However, cPAHs were rarely detected.[128] Thus, these cPAH data are not presented.

7.4.1 PCBs

A search was conducted to identify available PCB tissue data from background areas in the greater Puget Sound area. The first step included a general online search, including the Web sites for WDFW, the King County Department of Natural Resources (KCDNR), Ecology, and WSDOH. A search was also conducted of the EIM database on Ecology's Web site (Ecology 2007c). In addition, both WDFW and WSDOH were contacted to confirm that all available data had been located.

A summary of available tissue data is provided in Table 7-19. Total PCBs were calculated as the sum of detected Aroclors, unless total PCB concentrations were provided in the individual data sources, in which case the reported value for total PCBs was used. If no individual Aroclors were detected in a sample, the total PCB concentration was reported as undetected at the highest RL for an individual Aroclor.

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[127] The range of dioxin and furan TEQs in sediments of 11 lakes and reservoirs throughout the United States, which were selected to represent background conditions in areas removed from known sources, was 0.12 to 16.3 ng/kg dw (EPA 2000b), with an arithmetic mean of 5.3 ng/kg dw.

[128] The detection limit for the available background samples (King County 1999e) was higher than that achieved in the LDW, further reducing the benefit of using such samples to characterize background.

Table 7-19. Total PCB concentrations in fish, crab, and clam composite tissue collected from non-urban and near-urban areas of the Puget Sound

Species	Tissue Type	Sampling Location	Sampling Year(s)	Detection Frequency	Individuals per Composite (Average)	Total PCB Concentration (µg/kg ww)			Source
						Mean[a]	Minimum	Maximum	
Rockfish									
Copper rockfish	fillet	PSAMP – non-urban[b]	1989 – 1999	1/1	5	4.5	4.5 J	4.5 J	West et al. (2001)
Copper rockfish	fillet	PSAMP – near-urban[b]	1989 – 1999	17/17	3.6	9.9	3.9 J	22.5 J	West et al. (2001)
Quillback rockfish	fillet	PSAMP – non-urban[b]	1989 – 1999	42/79	2.6	12.2	2.0 J	124.4 J	West et al. (2001)
Quillback rockfish	fillet	PSAMP – near-urban[b]	1989 – 1999	42/42	2.4	40.3	4.1 J	140.4 J	West et al. (2001)
Yelloweye rockfish	fillet	PSAMP non-urban[b]	1989 – 1999	2/2	1	32.3	16.3 J	48.2 J	West et al. (2001)
Sole									
English sole	fillet	Case Inlet/ Dana Passage	2005	2/3	4.67	7.2	5.6 J	13.2 J	Era-Miller (2006)
English sole	fillet	Pickering Passage	2005	0/2	5	2.8	5.5 U	5.6 U	Era-Miller (2006)
English sole	fillet	South Puget Sound	2005	2/2	20	6.5	6.1 J	6.8 J	Era-Miller (2006)
English sole	fillet	PSAMP – non urban[b]	1989 – 1999	117/189	15.2	11.6	1.3 J	50.8 J	West et al. (2001)
English sole	fillet	PSAMP – near urban[b]	1989 – 1999	66/78	15.1	16.2	2.0 J	79.9 J	West et al. (2001)
English sole	na	Discovery Bay	1982	1/1	na	13	13	13	Gahler et al. (1982), as cited in cited in Cubbage (1992)
Flathead sole	na	Discovery Bay	1982	0/1	na	5	10 U	10 U	Gahler et al. (1982), as cited in cited in Cubbage (1992)
Rock sole	fillet	Budd Inlet	2005	0/1	5	2.7	5.4 U	5.4 U	Era-Miller (2006)
Rock sole	fillet	Carr Inlet	2005	0/2	5	2.3	5.5 U	5.5 U	Era-Miller (2006)
Rock sole	fillet	Case Inlet/ Dana Passage	2005	0/1	5	2.3	5.5 U	5.5 U	Era-Miller (2006)
Rock sole	fillet	Hale Passage	2005	0/2	5	2.7	5.1 U	5.5 U	Era-Miller (2006)

SPECIES	TISSUE TYPE	SAMPLING LOCATION	SAMPLING YEAR(S)	DETECTION FREQUENCY	INDIVIDUALS PER COMPOSITE (AVERAGE)	TOTAL PCB CONCENTRATION (µg/kg ww)			SOURCE
						MEAN[a]	MINIMUM	MAXIMUM	
Crab									
Dungeness crab	hepato-pancreas	Dungeness Bay[c]	2006	7/7	1	25.0	13.1	49.5	Malcolm Pirnie (2007a)[d]
Dungeness crab	hepato-pancreas	Freshwater Bay[c]	2006	8/8	1	17.8	8.80	32.3	Malcolm Pirnie (2007a)[d]
Dungeness crab	edible meat	Dungeness Bay[c]	2006	7/7	1	1.02	0.46	1.92	Malcolm Pirnie (2007a)[d]
Dungeness crab	edible meat	Freshwater Bay[c]	2006	8/8	1	0.62	0.43	0.99	Malcolm Pirnie (2007a)[d]
Dungeness crab	edible meat	Padilla/Fidalgo Bay	1999	1/3	5	1.2	1.2J	2.6U	Ecology (2000)
Dungeness crab	edible meat	Samish Island	1999	1/1	5	1.4	1.4J	1.4J	Ecology (2000)
Dungeness crab	calculated whole body[e]	Dungeness Bay[c]	2006	7/7	1	8.44	4.39	16.05	Malcolm Pirnie (2007a)[d]
Dungeness crab	calculated whole body[e]	Freshwater Bay[c]	2006	8/8	1	5.96	3.03	10.71	Malcolm Pirnie (2007a)[d]
Clams									
Geoduck	tissue	Freshwater Bay[c]	2006	8/8	1	0.64	0.24	1.43	Malcolm Pirnie (2007a)[d]
Geoduck	visceral cavity	Freshwater Bay[c]	2006	5/5	1	1.35	0.92	2.10	Malcolm Pirnie (2007a)[d]
Horse clam	tissue	Dungeness Bay[c]	2006	8/8	1	0.12	0.09	0.14	Malcolm Pirnie (2007a)[d]
Horse clam	tissue	Freshwater Bay[c]	2006	8/8	1	0.14	0.10	0.23	Malcolm Pirnie (2007a)[d]
Horse clam	visceral cavity	Dungeness Bay[c]	2006	5/5	1	1.06	0.35	1.49	Malcolm Pirnie (2007a)[d]
Horse clam	visceral cavity	Freshwater Bay[c]	2006	5/5	1	1.66	1.35	2.14	Malcolm Pirnie (2007a)[d]
Butter clam	soft parts	Samish Island	1999	0/1	50	1.25	2.5 U	2.5 U	Ecology (2000)
Butter clam	soft parts	various locations[f]	1994 – 1998	1/66	na	na	3.5[g]	3.5[g]	King County
Littleneck clam	soft parts	Padilla/Fidalgo Bay	1999	1/3	50	2.8	2.3 U	3.6	Ecology (2000)

[a] Mean concentrations were calculated using one-half the highest RL for an individual Aroclor for samples in which no Aroclors were detected.

[b] PSAMP data are from various sites around Puget Sound (Maps 7-9 through 7-11).

[c] Dungeness Bay and Freshwater Bay were the reference sites used in the Rayonier Mill RI near Port Angeles, Washington (Malcolm Pirnie 2007a).

[d] The total PCB concentrations in this study were calculated as the sum of all detected PCB congeners.

[e] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Total PCB concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

[f] Locations include Edmonds, Point Wells, Carkeek Park, Golden Gardens, West Point, Alki Point, Vashon Island, and Normandy Park. Data for clams collected by King County were compiled from five King County reports (1995, 2000e, 2001b, 2005j, 2006f).

[g] One detected concentration was reported for a single Aroclor (1254) in a single sample. Other Aroclors were not detected, and RLs were not reported for the other samples.

J – estimated concentration
na – not available
PCB – polychlorinated biphenyl
PSAMP – Puget Sound Ambient Monitoring Program
U – not detected
ww – wet weight

Tissue background numbers are presented in this section to provide context for the tissue RBTCs presented in Section 8. It should be noted that sampling and analytical methods used to produce these datasets varied from study to study. Thus, while these data provide a general indication of chemical concentrations in these reference areas, they should not be viewed as a single dataset with consistent methodology. Furthermore, because of differences in methods and the potential for spatial bias,[129] averages or UCL concentrations on a species-specific tissue basis across all background locations were not derived. The results of various studies are presented only for informational purposes.

Background tissue data were obtained from numerous locations around Puget Sound (Maps 7-9, 7-10, and 7-11). The primary sources of background data for fish were a PSAMP report (West et al. 2001) and a study conducted by Ecology (Era-Miller 2006). The PSAMP study area encompassed the Puget Sound region from Georgia Basin to the South Sound, including over 2,300 km^2 of aquatic habitat and over 2,100 km of shoreline (West et al. 2001) (Maps 7-9 and 7-10). Locations throughout the study area were sampled intermittently over a 10-year period from 1989 through 1999 such that all locations were sampled at least once, but not all locations were sampled every year. The species analyzed included English sole, copper rockfish, quillback rockfish, yelloweye rockfish, and adult salmon[130] (both chinook and coho species).

The study conducted by Ecology (Era-Miller 2006) was designed to augment the PSAMP data, and thus some of the locations were co-located with those in the PSAMP monitoring effort. The sampling locations included eight sites in south Puget Sound (Map 7-9). The species sampled during this investigation were English sole and rock sole (see Table 7-19). Limited information on two fish samples was also found in a report by Cubbage (1992) regarding a study conducted by Gahler et al. (1982). This study presented total PCB data from samples collected in Discovery Bay.

The primary source of crab and clam data was the remedial investigation (RI) for the Rayonier mill site near Port Angeles (Malcolm Pirnie 2007a). This investigation included tissue sampling at Freshwater and Dungeness Bays; the locations were selected to represent background conditions for that RI (Map 7-11). Species collected in background locations for the Rayonier Mill RI included Dungeness crabs, horse clams, and geoducks.

As part of another study, conducted by Ecology (2000), shellfish tissue samples, including Dungeness crabs, butter clams, and littleneck clams, were collected and

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[129] An example of spatial bias is one location having a larger number of samples compared to another location.

[130] Adult salmon data were not included in Table 7-5 because their exposure to chemicals in LDW sediment is not anticipated to significantly influence the concentrations in their tissues, primarily because of the very small portion of their lives spent in the LDW.

analyzed. In addition, King County analyzed shellfish samples for PCBs as part of their water quality assessment surveys (King County 1995, 2000e, 2001b, 2005j, 2006f). Out of the 66 samples collected by King County from various locations around Puget Sound (including Edmonds, Point Wells, Carkeek Park, Golden Gardens, West Point, Alki Point, Vashon Island, Normandy Park), PCBs (Aroclor 1254) were detected in only one sample.

In general, total PCB concentrations ranged from non-detect to 140.4 µg/kg ww in fish tissue samples (Table 7-19; Figure 7-13). Rock sole had the lowest total PCB concentrations, followed by copper rockfish, English sole, yelloweye rockfish, and quillback rockfish. Total PCB concentrations in whole-body Dungeness crabs ranged from 3.03 to 16.05 µg/kg ww. For all species for which both near and non-urban data were available, the non-urban samples consistently had lower mean total PCB concentrations than near-urban samples. Total PCB concentrations in clam tissue samples ranged from non-detect to 3.6 µg/kg ww, and thus were the lowest of the background tissue types measured. Total PCB concentrations in shellfish tissue ranged from non-detect to 3.6 µg/kg ww.

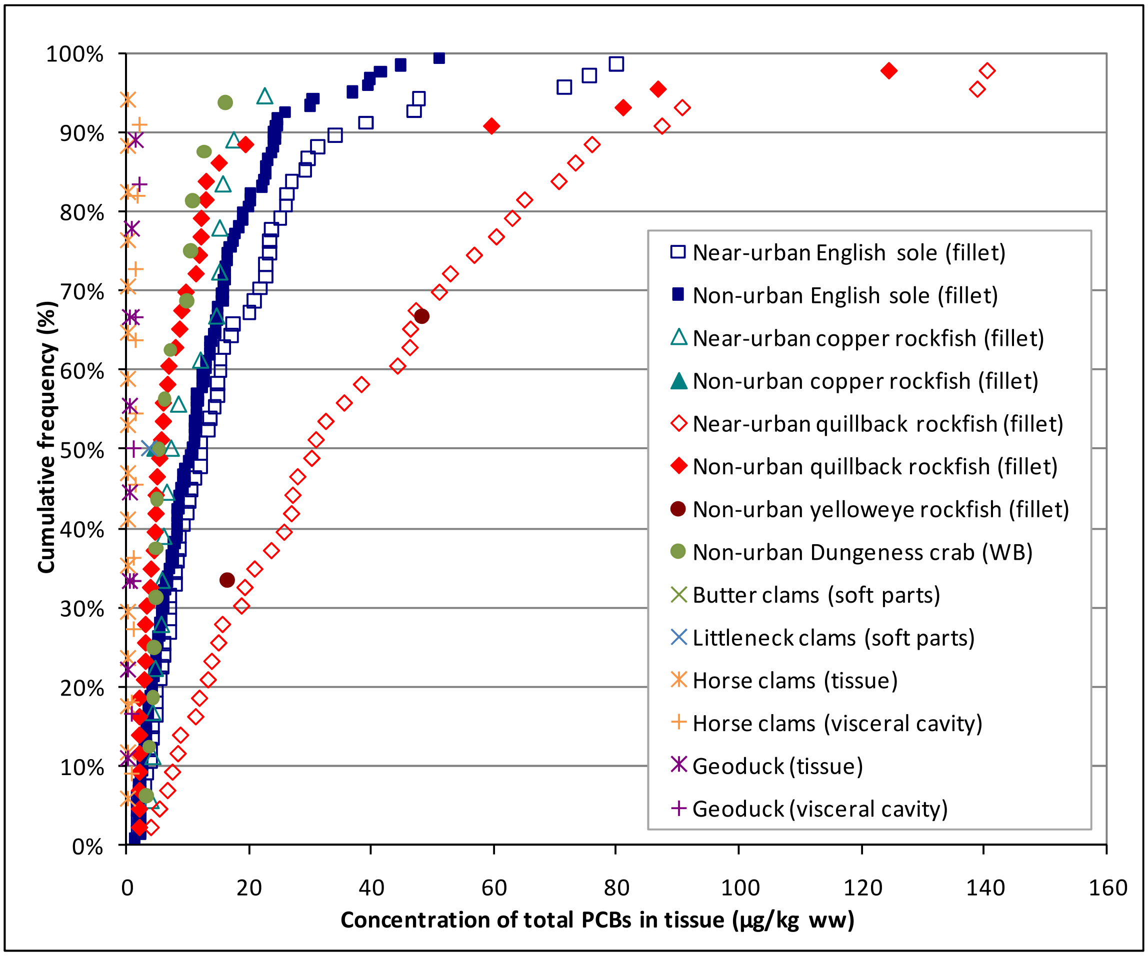

Figure 7-13. Detected total PCB concentrations in tissue samples collected at non-urban and near-urban locations in greater Puget Sound

PCBs have been detected in many types of food other than seafood, such as butter, chicken, steak, eggs and baked goods (WSDOH 2006). The concentrations reported in supermarket foods were similar to those detected in background fish samples, and ranged from non-detect to 70 µg/kg ww.

7.4.2 Arsenic

Data were collected as part of the RI to assess background concentrations of arsenic (both total and inorganic) in tissue (Windward 2005a, b, 2006b). Samples were collected from locations both within and outside of the region thought to be impacted by aerial emissions from the former Asarco smelter in Tacoma, Washington (Area-Wide Soil Contamination Task Force 2003), as noted in Table 7-20 and shown on Map 7-12. This smelter is known have been an important source of arsenic to areas within the smelter plume and has been the focus of an extensive soil study conducted by Ecology in south King County (Ecology 2001b).

The concentrations of inorganic arsenic are of greater interest because inorganic arsenic is more toxic and because EPA has derived a cancer slope factor for inorganic arsenic but not for total or organic arsenic. Therefore, only inorganic arsenic data are summarized. Clams were collected from background areas in both 2004 and 2005 and fish and crabs were collected in 2004 as part of the RI (Table 7-20 and Map 7-12). The only other study in Puget Sound that analyzed inorganic arsenic in tissue was conducted by Ecology (2002a), but the results from that study were not relevant because samples were collected only from urban bays.

In 2004, fish and crabs were collected from the East Passage (Asarco-influenced) and Blake Island (non-Asarco-influenced) and combined to create 61 composite tissue samples, as presented in Table 7-20 (Windward 2005b). The 2004 tissue dataset also included 12 composite clam tissue samples, 6 from a beach in Seahurst Park (Asarco-influenced) and 6 from a beach on Bainbridge Island (non-Asarco-influenced) (Windward 2005a). Because the clams collected in 2004 were not *Mya arenaria* (the type of clam predominantly found in the LDW) further sampling was conducted in 2005 to collect *Mya arenaria* from other background locations for the analysis of inorganic and total arsenic. In the 2005 sampling event, 12 additional composite *Mya arenaria* samples were formed, 6 from beaches on Vashon Island (Asarco-influenced) and 6 from the Dungeness National Wildlife Refuge (NWR) (non-Asarco-influenced) (Windward 2006).

Table 7-20. Inorganic arsenic concentrations in clam, fish, and crab tissue composite samples collected from Puget Sound locations outside of urban areas

AREA	TISSUE TYPE	SAMPLING YEAR	DETECTION FREQUENCY	INORGANIC ARSENIC (mg/kg ww)			SOURCE
				MEAN[a]	MINIMUM	MAXIMUM	
Clams							
Vashon Island (Asarco influenced) – *Mya arenaria*	whole body	2005	6/6	0.143	0.0930	0.227	Windward (2006b)
Seahurst Park (Asarco influenced) – various species[b]	whole body	2004	6/6	0.443	0.0980 J	0.616 J	Windward (2005a)
Dungeness NWR (non-Asarco influenced) – *Mya arenaria*	whole body	2005	6/6	0.0645	0.0470	0.117	Windward (2006b)
Bainbridge Island (non-Asarco influenced) – various species[b]	whole body	2004	6/6	0.201	0.0440 J	0.485 J	Windward (2005a)
Perch (shiner surfperch, pile perch, and striped perch)							
East Passage (Asarco influenced)	whole body	2004	2/3	0.008	0.005 U	0.01 J	Windward (2005b)
Blake Island (non-Asarco influenced)	whole body	2004	6/6	0.02	0.01	0.03	Windward (2005b)
Crabs (Dungeness crab and slender crab)							
East Passage (Asarco influenced)	edible meat	2004	6/6	0.018	0.010 J	0.040	Windward (2005b)
	hepatopancreas	2004	2/2	0.08	0.08	0.08	Windward (2005b)
	calculated whole body[c]	2004	6/6	0.037	0.032 J	0.052	Windward (2005b)
Blake Island (non-Asarco influenced)	edible meat	2004	6/6	0.023	0.020	0.030	Windward (2005b)
	hepatopancreas	2004	2/2	0.31	0.27	0.34	Windward (2005b)
	calculated whole body[c]	2004	6/6	0.11	0.10	0.13	Windward (2005b)
Benthic fish (English sole and starry flounder)							
East Passage (Asarco influenced)	fillet	2004	1/6	0.002	0.002 U	0.004 J	Windward (2005b)
	whole body	2004	6/6	0.01	0.007 J	0.02	Windward (2005b)
Blake Island (non-Asarco influenced)	fillet	2004	2/6	0.003	0.002 U	0.005 U	Windward (2005b)
	whole body	2004	6/6	0.02	0.01	0.03	Windward (2005b)

[a] Means were based on one-half RL for non-detect.

[b] Composite clam tissue samples from Seahurst Park and Bainbridge Island included multiple species (*Saxidomus giganteus, Clinocardium nuttallii, Macoma nasuta, Macoma secta, Tresus capax*, and *Protothaca staminea*).

[c] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Arsenic concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

NWR – National Wildlife Refuge

J – estimated concentration

U – not detected

ww – wet weight

Inorganic arsenic concentrations in clam, fish, and crab tissue samples ranged from non-detect to 0.616 mg/kg ww. Inorganic arsenic concentrations were generally higher in clams (0.0440 to 0.616 mg/kg ww) than in the other tissue types (non-detect to 0.34 mg/kg ww). In clams, most of the inorganic arsenic concentrations in Asarco-influenced samples were higher than concentrations in non-Asarco-influenced samples, although there was some overlap in concentrations and two samples from Bainbridge Island (non-Asarco-influenced) had relatively high concentrations (Figure 7-14). No statistical analyses were conducted to evaluate whether the difference was significant.

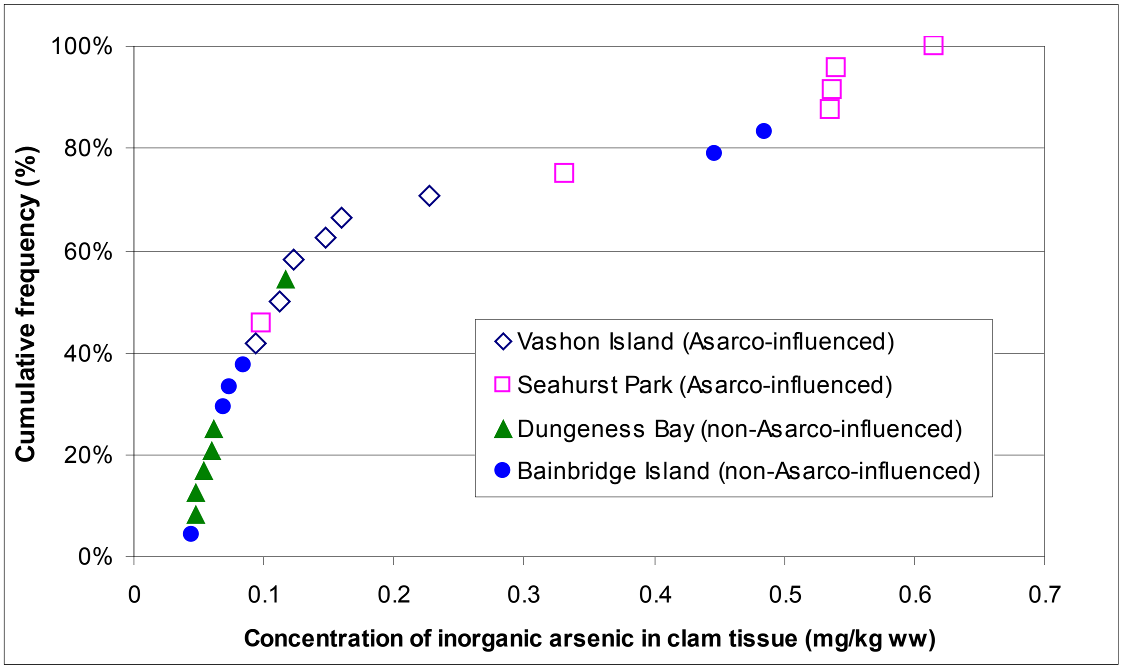

Figure 7-14. Comparison of inorganic arsenic concentrations in clam tissues collected from Puget Sound

7.5 SUMMARY

This section presents an initial summary of data for total PCBs, arsenic, cPAHs, and dioxins and furans from areas outside of the LDW and for cores collected from the Upper Turning Basin of the LDW. These data provide context for information presented elsewhere in the RI and will be useful in future background determinations. These datasets include surface sediment data from Puget Sound reference areas and other recent data that are potentially relevant for natural background concentrations; data from the Green River (upstream of the LDW), including surface sediment concentrations and TSS-normalized concentrations in water; data from sediment core samples from the Upper Turning Basin; and tissue data for various species collected from the greater Puget Sound region. Information on urban sediment concentrations from multiple areas is also presented to provide additional context.

Individual datasets address some or all of the four primary chemicals for which background values are of interest: PCBs, arsenic, cPAHs, and dioxins and furans. Detailed evaluations of background datasets and determinations of background-based PRGs will be addressed in continuing assessments to be documented in the FS or separate technical memoranda. Additional sampling studies, which are providing new data relevant to background, are in progress and will be considered in future evaluations. The information presented here is therefore preliminary.

Although single "point" concentrations may be used to summarize potential background datasets, as is done in this summary, background is properly viewed as a statistical distribution of values. A final statistical approach for determining background values for cleanup decisions, or for evaluating compliance with background-based cleanup levels, has not yet been selected. Statistical evaluations within this section are limited to estimating simple summary statistics. Some data representativeness issues are mentioned as datasets are described, primarily to illustrate some of the factors to be reviewed, but with no attempt to be comprehensive. Both statistical and data representativeness issues will be further evaluated as background assessments continue.

Concentrations of PCBs, arsenic, cPAHs, and dioxins and furans in surface sediment were summarized for reference areas throughout Puget Sound as an indication of the concentrations of these chemicals in non-urban areas. Specific natural background concentrations or concentration ranges were not derived; these concentrations may be developed in the future by EPA and Ecology. Data from the Puget Sound 2008 survey conducted by the DMMP became available in January 2009. That study provided additional data for PCB congeners and dioxins and furans.

Based on currently available data, the estimated 90th percentile concentrations[131] from each of the reference areas and the Puget Sound 2008 survey ranged from 2.3 to 15.9 mg/kg dw for arsenic, from 0.2 to 19.9 µg/kg dw for PCBs, from 14.7 to 244.7 µg/kg dw for cPAHs, and from 0.141 to 2.3 ng/kg dw for dioxin and furan TEQs.

Sediment and water quality data from upstream areas and core data from the LDW Upper Turning Basin were compiled to assess the range of chemical concentrations in sediments entering the LDW from upstream. Table 7-7 summarizes the means, medians, 90th percentiles, and UCLs for each dataset that was evaluated. For PCBs, the range of 90th percentile concentrations was 40 to 107 µg/kg dw. The lowest 90th percentile was associated with relatively coarse-grained surface sediment samples collected upstream of RM 5.0. The highest 90th percentile concentration was an estimated concentration associated with fine suspended solids in the water column upstream of the site. The upstream 90th percentile concentrations were within the

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[131] The 90th percentile concentration ranges are presented as one point of comparison only for informational purposes. Estimated 90th percentile values that reflect non-detected results have been replaced with maximum detected values.

range but generally lower than the 90th percentile values for sediments in urban bays and lakes, which ranged from 64 to 217 µg/kg dw, with the exclusion of the inner Elliott Bay data.

For arsenic, the 90th percentile concentrations in sediment calculated from the individual datasets ranged from 11 to 73 mg/kg dw. The lowest 90th percentile concentration was associated with relatively coarse-grained surface sediment samples collected upstream of the site, whereas the highest 90th percentile concentration was an estimated concentration associated with fine suspended solids in the water column upstream of the site. The 90th percentile arsenic concentrations for sediments in urban bays and lakes ranged from 9.8 to 38 mg/kg dw.

For cPAHs, the 90th percentile concentrations in sediment calculated for the individual datasets ranged from 135 to 354 µg/kg dw, with the lowest 90th percentile concentration associated with relatively coarse-grained surface sediment samples collected upstream of RM 5.0 and the highest 90th percentile concentration associated with suspended solids from upstream of the LDW. The 90th percentile cPAH concentration for sediments in urban bays and lakes ranged from 185 to 904 µg/kg dw, with the exclusion of the inner Elliott Bay data.

There were fewer data available for dioxins and furans. The mean dioxin and furan TEQ from sediment upstream of RM 5.0 was 2.0 ng/kg dw, which was lower than the 90th percentile dioxin and furan TEQ for samples collected near storm drains and other areas receiving runoff from the greater Seattle metropolitan area (16.3 ng/kg dw). Because only four samples were available from upstream, no 90th percentile value was estimated for dioxins and furans.

Total PCB and inorganic arsenic concentrations in tissue samples collected outside of urban areas were presented to provide a context for PCB and inorganic arsenic tissue RBTCs presented in Section 8. Total PCB concentrations in fish fillets and crab tissue ranged from non-detect to 140.4 µg/kg ww among the species. Total PCB concentrations in clams were generally lower than in fish and crabs, ranging from non-detect to 3.6 µg/kg ww. Inorganic arsenic concentrations in clam tissue ranged from 0.0440 to 0.616 mg/kg ww. Inorganic arsenic concentrations in fish and crabs (whole body) were generally lower (non-detect to 0.13 mg/kg ww).

EPA and Ecology will set final background concentrations for consideration of cleanup levels in the ROD. The preliminary information on background presented in this section, based on available datasets, will be supplemented by additional datasets as they become available. Further detailed evaluations of background are planned. This initial compilation of potential background data provides a context for the LDW site characterization data (Section 4) and RBTCs (Section 8).

8 Risk-Based Threshold Concentrations

This section presents the derivation of RBTCs, which represent the concentrations that equate to specific risk thresholds. RBTCs for human health were developed for excess cancer risk thresholds of 1×10^{-4}, 1×10^{-5} and 1×10^{-6} and HQs equal to 1. These thresholds are important in both the CERCLA and MTCA programs, as discussed in the HHRA (Appendix B). RBTCs have been estimated for exposure through direct sediment contact and ingestion of seafood items. RBTCs were developed for chemicals that were identified as risk driver chemicals in either the ERA (Appendix A) or the HHRA (Appendix B). RBTCs are an important component in the derivation of PRGs in the FS. The RBTCs derived in this section are not the same as cleanup levels or cleanup goals. EPA and Ecology will select final cleanup values in the ROD. It should also be noted that although SWACs are discussed in this section, PRGs will be compared with 95% UCLs on the SWACs in the FS.

In the HHRA (Appendix B, Section B.7), PCBs, arsenic, cPAHs, and dioxins and furans were identified as risk driver chemicals for both seafood consumption and direct sediment contact scenarios (Table 8-1). In the ERA (Appendix A, Section A.7), PCBs were identified as a risk driver chemical for river otter based on risks associated with ingestion of PCBs in prey.[132] In addition, 41 chemicals were identified as risk driver chemicals for the benthic invertebrate community based on detected exceedances of SMS criteria.

Table 8-1. Summary of risk driver chemicals from the HHRA and ERA

EXPOSURE SCENARIO	PCBS	ARSENIC	DIOXINS AND FURANS	cPAHS	OTHER CHEMICALS
Human health – direct sediment contact	X	X	X	X	ns
Human health – seafood consumption	X	X	X[a]	X	ns
River otter prey ingestion	X	na[b]	ne	na[c]	na[b]
Benthic invertebrate sediment exposure	X	X	ne	na[c]	X[d]

[a] Dioxins and furans were assumed to be risk drivers for human health seafood consumption (see Appendix B) even though tissue data were not available for the risk assessment. Because tissue data and risk estimates are not available, RBTCs cannot be calculated for dioxins and furans for the seafood consumption scenarios.

[b] Arsenic and other chemicals were not identified as risk drivers because they were not COCs for otter.

[c] cPAHs (as a TEQ) were evaluated only for human health risks. Other approaches (e.g., assessment of individual PAH compounds) were used in the ecological assessment of PAHs.

[d] In addition to arsenic and total PCBs, risk driver chemicals identified for the benthic invertebrate community, based on one or more surface sediment samples with exceedances of the SQS, include cadmium, chromium, copper, lead, mercury, silver, zinc, acenaphthene, anthracene, benz(a)anthracene, benzo(a)pyrene,

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[132] In the ERA (Appendix A), direct exposure to sediments was explicitly evaluated for fish and wildlife receptors as a component of the overall diet (i.e., incidental sediment ingestion). No risk drivers were identified for fish or wildlife receptors based solely on direct contact with sediments because the majority of the risk is associated with ingestion of dietary items, not direct contact with sediment. Therefore, the PCB RBTC for river otter is based only on ingestion of prey items.

benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3,-c,d)pyrene, naphthalene, phenanthrene, pyrene, total benzofluoranthenes, HPAH, LPAH, BEHP, butyl benzyl phthalate, dimethyl phthalate, 1,2-dichlorobenzene, 1,4-dichlorobenzene, 1,2,4-trichlorobenzene, 2-methylnaphthalene, 4-methylphenol, 2,4-dimethylphenol, benzoic acid, benzyl alcohol, dibenzofuran, hexachlorobenzene, n-nitrosodiphenylamine, pentachlorophenol, and phenol.

BEHP – bis(2-ethylhexyl) phthalate
cPAH – carcinogenic polycyclic aromatic hydrocarbon
COC – chemical of concern
ERA – ecological risk assessment
HHRA – human health risk assessment
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
na – not applicable
ne – not evaluated (in risk assessment)
ns – not selected (as a risk driver)
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SQS – sediment quality standards
TEQ – toxic equivalent

Various approaches were used to derive the RBTCs for the risk driver chemicals in Table 8-1. Mathematical equations based on risk assessment assumptions were used to derive sediment RBTCs for the human health direct sediment contact scenarios (Section 8.1) and tissue RBTCs for the seafood ingestion scenarios (human [Section 8.2] and otter [Section 8.4]). Sediment RBTCs for the benthic invertebrate community were based on SMS criteria (Section 8.5).

To develop RBTCs for sediment for seafood consumption scenarios, a relationship was needed between chemicals concentrations in tissues and sediment. Different types of models were used to estimate this relationship, depending on data type. For PCBs, a FWM was developed to relate total PCB concentrations in sediment to concentrations in fish, crabs, and clams, which were all relevant for RBTC development (Section 8.3.1). Co-located sediment and clam tissue data were used to evaluate regression models for arsenic and cPAHs to determine if the derivation of sediment RBTCs was appropriate (Sections 8.3.2 and 8.3.3).

The FWM developed for PCBs could not be used to relate arsenic concentrations in clam tissue and sediment and was not used to relate cPAHs in sediment and clam tissue. The FWM could not be used for arsenic because the Gobas-type FWM was designed to model bioaccumulation of hydrophobic organic chemicals. The FWM was not used for cPAHs because application of the model to a toxicity equivalent mixture of compounds is complicated, and would have required certain input parameters that are not readily available (such as a K_{OW} value weighted to account for differences in toxicity of the various cPAH congeners and cPAH concentrations in water). Also, the FWM was not calibrated for PAH compounds.

8.1 SEDIMENT RBTCS FOR HUMAN DIRECT SEDIMENT CONTACT

Sediment RBTCs were determined for direct sediment contact scenarios evaluated in the HHRA (Appendix B). RBTCs are important in the evaluation of remedial action objectives (RAOs); RBTCs developed for RME scenarios will be used in evaluating cleanup alternatives.[133] RBTCs for non-RME direct sediment contact scenarios were also estimated to aid in risk management and risk communication.

The RME scenarios for direct sediment contact evaluated in the HHRA (Appendix B) included beach play, netfishing, and tribal clamming (120 days per year). Non-RME scenarios included netfishing (central tendency) and clamming scenarios with lower (i.e., 7 days per year, provided for informational purposes) and higher (i.e., 183 days per year) exposure frequencies than the RME clamming scenario. Key parameters used to estimate risks associated with direct sediment contact are summarized in Tables 8-2 and 8-3. Detailed descriptions of the direct sediment contact scenarios and risk results are presented in the HHRA (Appendix B) in Sections B.3 and B.5, respectively. Detailed exposure information for the direct sediment contact scenarios is presented in Tables B.3-15 through B.3-26, EPCs are summarized in Table B.3-44, chemical toxicity information is presented in Tables B.4-1 and B.4-2, and risk estimates for direct sediment contact scenarios are presented in Tables B.5-23 through B.5-48.

Table 8-2. Summary of input parameters for RME direct sediment contact exposure scenarios

Direct Sediment Contact Scenario for Which RBTCs Were Derived	Incidental Sediment IR (g/day)	Exposure Frequency (days/year)	Exposure Duration (years)	Skin Surface Area Exposed (cm^2)
Netfishing RME	0.05	119	44	3,600
Child beach play RME[a]	0.2	65	6	varies with age
Tribal clamming RME (120 days per year)	0.1	120	64	6,040

[a] For the beach play scenarios, the LDW was divided into eight sections to assess risks associated with different areas of the LDW. Exposure of children was evaluated from the time of birth to 6 years of age.

IR – ingestion rate

RBTC – risk-based threshold concentration

RME – reasonable maximum exposure

Table 8-3. Summary of input parameters for non-RME direct sediment contact exposure scenarios

Direct Sediment Contact Scenario for Which RBTCs Were Derived	Incidental Sediment IR (g/day)	Exposure Frequency (days/year)	Exposure Duration (years)	Skin Surface Area Exposed (cm^2)
Netfishing CT	0.05	63	29	3,600
Clamming (7 days per year)	0.1	7	30	6,040
Tribal clamming upper bound (183 days per year)	0.1	183	70	6,040

CT – central tendency

IR – ingestion rate

RBTC – risk-based threshold concentration

RME – reasonable maximum exposure

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[133] In accordance with EPA guidance (Section 6.1.2 of EPA 1989), RME scenarios are used to formulate RAOs and evaluate cleanup alternatives at Superfund sites.

8.1.1 Methods for calculating sediment RBTCs for human direct sediment contact

Sediment RBTCs for direct sediment contact scenarios were derived for each of the risk driver chemicals according to Equation 8-1 (for carcinogenic effects). The first term in the denominator addresses dermal exposure; the second term addresses incidental ingestion.

$$\text{RBTC} = \frac{\text{TR}}{\left[\left[\left(\frac{\text{ABS} \times \text{SA} \times \text{AF} \times \text{FC} \times \text{EF} \times \text{ED} \times \text{CF}_1}{\text{BW} \times \text{AT}_\text{C}}\right) + \left(\frac{\text{IR} \times \text{FC} \times \text{EF} \times \text{ED} \times \text{CF}_2}{\text{BW} \times \text{AT}_\text{C}}\right)\right] \times \text{SF}\right]} \qquad \textbf{Equation 8-1}$$

Where:

RBTC = risk-based threshold concentration in sediment (mg/kg dw)
TR = target risk (i.e., 1×10^{-6}, 1×10^{-5}, 1×10^{-4}) (unitless)
ABS = absorption factor (unitless)
SA = skin surface area (cm^2)
AF = adherence factor (mg/cm^2-day)
FC = fraction from contaminated site (unitless)
EF = exposure frequency (days/year)
ED = exposure duration (years)
CF_1 = conversion factor, kg to mg (0.000001)
BW = body weight (kg)
AT_c = averaging time, cancer (days)
SF = cancer slope factor (mg/kg-day)$^{-1}$
AT_n = averaging time, non-cancer (days)
IR = incidental sediment ingestion rate (g/day)
CF_2 = conversion factor, kg to g (0.001)

RBTCs were not estimated for non-cancer hazards for direct-contact scenarios because none of the RME scenarios had HQs for an individual chemical greater than 1 or generated endpoint-specific HIs in excess of 1 (Appendix B, Section B.5.6). Equation 8-1 is simply a rearrangement of the equation used in the HHRA to estimate excess cancer risk. In this case, rather than inserting a maximum or 95% UCL sediment concentration into the equation and solving for risk, as was done in the HHRA (Appendix B), the target risk is set and the equation is solved for a corresponding sediment concentration.

8.1.2 Results of sediment RBTC calculations for human direct sediment contact

Sediment RBTCs for the direct sediment contact RME scenarios are shown in Table 8-4; those for non-RME scenarios are presented in Table 8-5. The non-RME RBTCs are presented for informational purposes only, and are not intended for use in setting clean up levels. The remedial decision-making process will focus primarily on the RME scenarios, although some non-RME scenarios will also be considered as lower- and upper- bound estimates of risk. The RBTCs for the tribal clamming RME scenario were lower than the RBTCs for the other two RME scenarios at a given target risk level, except for cPAHs. The beach play RME RBTCs were lowest for cPAHs because the beach play scenario was specifically designed to be protective of children. Because cPAHs have a mutagenic mode of action, EPA guidance (2005e) recommends that the dose estimates be adjusted upwards in the risk calculation to account for the potential greater susceptibility of children from 0 to 6 years of age compared with older children and adults. This approach resulted in approximately a five-fold increase in the cancer risk estimate for cPAHs for children (Appendix B, Section B.5, Equation 5.2), with a corresponding lowering of the cPAH RBTC. Considerations relevant to the application of these RBTCs are discussed in Section 8.6.

Table 8-4. Sediment RBTCs for RME direct sediment contact human health scenarios

RISK DRIVER	TARGET RISK	SEDIMENT RBTCS BASED ON DIRECT-CONTACT RME SCENARIOS		
		NETFISHING RME	BEACH PLAY RME	TRIBAL CLAMMING RME (120 days/year)
Arsenic (mg/kg dw)	1×10^{-6}	3.7	2.8	1.3
	1×10^{-5}	37	28	13
	1×10^{-4}	370	280	130
cPAHs (µg/kg dw)[a]	1×10^{-6}	380	90	150
	1×10^{-5}	3,800	900	1,500
	1×10^{-4}	38,000	9,000	15,000
Dioxin/furan TEQ (ng/kg dw)	1×10^{-6}	37	28	13
	1×10^{-5}	370	280	130
	1×10^{-4}	3,700	2,800	1,300
Total PCBs (µg/kg dw)	1×10^{-6}	1,300	1,700	500
	1×10^{-5}	13,000	17,000	5,000
	1×10^{-4}	130,000	170,000	50,000

[a] cPAH concentrations are expressed in terms of benzo(a)pyrene equivalents. Because of the potential for increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (2005e), the risk estimate for children for cPAHs is based on dose adjustments across the 0-to-6-year age range of children (see Appendix B, Section B.5.1, for more information).

cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
dw – dry weight
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure
TEQ – toxic equivalent

Table 8-5. Sediment RBTCs for non-RME direct sediment contact human health scenarios

RISK DRIVER	TARGET RISK	SEDIMENT RBTCS BASED ON NON-RME DIRECT-CONTACT SCENARIOS		
		CLAMMING (7 days/year)	NETFISHING CT	CLAMMING (183 days/year)
Arsenic (mg/kg dw)	1 × 10^{-6}	43	11	0.80
	1 × 10^{-5}	430	110	8.0
	1 × 10^{-4}	4,300	1,100	80
cPAHs (µg/kg dw)[a]	1 × 10^{-6}	4,700	1,100	87
	1 × 10^{-5}	47,000	11,000	870
	1 × 10^{-4}	470,000	110,000	8,700
Dioxin/furan TEQ (ng/kg dw)	1 × 10^{-6}	220	110	7.98
	1 × 10^{-5}	2,200	1,100	79.8
	1 × 10^{-4}	22,000	11,000	798
Total PCBs (µg/kg dw)	1 × 10^{-6}	16,000	9,500	300
	1 × 10^{-5}	160,000	95,000	3,000
	1 × 10^{-4}	1,600,000	950,000	30,000

[a] cPAH concentrations are expressed in terms of benzo(a)pyrene equivalents.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
dw – dry weight
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure
TEQ – toxic equivalent

8.2 TISSUE RBTCS FOR HUMAN SEAFOOD CONSUMPTION

Several different human seafood consumption scenarios were evaluated in the HHRA (Appendix B). EPA's application of the tribal framework to the LDW states that the tribal scenarios based on Tulalip data are the most appropriate scenarios for representing RME exposure in the LDW because of the quality and quantity of shellfish habitat that will be present post–remediation (EPA 2005a, 2007d). Therefore, RBTCs associated with the tribal adult and child RME seafood consumption scenarios based on Tulalip consumption data will be used in the evaluation of cleanup alternatives based on both the application of the tribal framework(EPA 2005a) and more recent communication from EPA (2007b). Because it provides another RME estimate, the adult API RME seafood consumption scenario may also be used in the evaluation of cleanup alternatives.

In addition to the RBTCs for the RME scenarios, RBTCs were developed for the other seafood consumption scenarios to aid in risk management and risk communication efforts. Thus, RBTCs were derived for adult tribal (Suquamish data), adult and child tribal CT (Tulalip data), adult API CT, and adult one-meal-per-month scenarios.

Key assumptions for each of the seafood consumption scenarios are summarized in Table 8-6. The seafood consumption scenarios and risk results are presented in the HHRA (Appendix B), in Sections B.3 and B.5, respectively. Detailed exposure information for the seafood consumption scenarios is presented in Tables B.3-7 through B.3-14, EPCs are summarized in Table B.3-39, chemical toxicity information is presented in Tables B.4-1 and B.4-2, and risk estimates are presented in Tables B.5-1 through B.5-16.

Table 8-6. Summary of key assumptions in the seafood consumption scenarios

Seafood Consumption Scenario for which RBTCs Were Derived	Ingestion Rate (g/day)					Exposure Duration (yrs)
	Pelagic Fish	Benthic Fish	Crabs	Other Shellfish[a]	Total	
Adult tribal RME (Tulalip data)	8.1	7.5	37.7[c]	44.2[c]	97.5	70
Child tribal RME (Tulalip data)	3.2	3.0	15.1[c]	17.7[c]	39.0	6
Adult API RME	4.9	2.4	10.6	33.6	51.5	30
Adult tribal CT (Tulalip data)	1.3	1.2	5.8[c]	6.7[c]	15.0	30
Child tribal CT (Tulalip data)	0.52	0.48	2.4[c]	2.64[c]	6.0	6
Adult tribal (Suquamish data)	56	29.2[d]	55[d]	444[d]	584.2[d]	70
Adult API CT	0.5	0.24	1.1	3.5	5.3	12
Adult one meal per month[b]	7.5	7.5	7.5	7.5	na	30

[a] Other shellfish represents both clam and mussel consumption categories.

[b] Adult one-meal-per-month consumption was evaluated for individual seafood categories independently to reflect different seafood consumption practices.

[c] Consumption rates based on the Tulalip Tribes survey (Toy et al. 1996) have been updated according to the correction provided by EPA (2009), in which the relative apportionment of shellfish between crab and other shellfish was reversed (although the total consumption did not change). These values differ from those presented in Table 6-2 (which contains the values previously provided by EPA and used in the LDW HHRA, Appendix B).

[d] Totals of benthic fish and shellfish categories (and overall total) differ slightly from values provided by EPA (2007d) because of significant figure and rounding issues when total consumption was allocated to more specific consumption categories. The overall total listed here is slightly higher than the corresponding value provided by EPA (2007d).

API – Asian and Pacific Islander

CT – central tendency

na – not applicable

RBTC – risk-based threshold concentration

RME – reasonable maximum exposure

Tissue RBTCs were estimated for total PCBs, arsenic, and cPAHs according to the methods described in Section 8.2.1. The primary use of these RBTCs will be to evaluate post-cleanup monitoring data for risk driver chemicals in seafood tissue. It should be noted that sediment RBTCs were not derived directly from tissue RBTCs; sediment RBTCs were derived as described in Section 8.3. Tissue RBTCs were not estimated for dioxins and furans because tissue data for these chemicals have not been collected from the LDW.

8.2.1 Methods for calculating tissue RBTCs for human seafood consumption

Tissue RBTCs associated with the human seafood consumption scenarios were estimated for total PCBs, arsenic, and cPAHs using Equation 8-2 (for carcinogenic effects) or Equation 8-3 (for non-carcinogenic effects; only PCBs and arsenic). The values for these parameters are presented in the HHRA (Section 3, Appendix B).

$$\text{Tissue RBTC} = \frac{\text{TR}}{\left[\left[\dfrac{\text{IR}\times\text{FC}\times\text{EF}\times\text{ED}\times\text{CF}}{\text{BW}\times\text{AT}_\text{C}}\right]\times\text{SF}\right]} \qquad \textbf{Equation 8-2}$$

$$\text{Tissue RBTC} = \frac{\text{THQ}}{\left[\left[\dfrac{\text{IR}\times\text{FC}\times\text{EF}\times\text{ED}\times\text{CF}}{\text{BW}\times\text{AT}_\text{n}}\right]\times\dfrac{1}{\text{RfD}}\right]} \qquad \textbf{Equation 8-3}$$

Where:

RBTC = risk-based threshold concentration in seafood items (mg/kg ww)
TR = target risk (i.e., 1×10^{-6}, 1×10^{-5}, 1×10^{-4}) (unitless)
IR = ingestion rate (g/day)
FC = fraction from contaminated site (unitless)
EF = exposure frequency (days/year)
ED = exposure duration (years)
CF = conversion factor, kg to g (0.001)
BW = body weight (kg)
AT_c = averaging time, cancer (days)
SF = cancer slope factor $(\text{mg/kg-day})^{-1}$
THQ = target hazard quotient (1) (unitless)
AT_n = averaging time, non-cancer (days)
RfD = reference dose (mg/kg-day)

RBTCs based on carcinogenic risks were estimated for three risk threshold levels, 1×10^{-6}, 1×10^{-5}, and 1×10^{-4}; for non-cancer hazards, the RBTCs corresponding to an HQ of 1 were estimated.

8.2.2 Results of tissue RBTC calculations for human seafood consumption

Tissue RBTCs corresponding to the RME seafood consumption scenarios are presented in Table 8-7; tissue RBTCs corresponding to the non-RME scenarios are presented in Table 8-8. RBTCs for non-RME scenarios are presented only for informational purposes. The remedial decision-making process will focus primarily on the RME scenarios, although some non-RME scenarios will also be considered as lower- and upper- bound estimates of risk. Tissue RBTCs for the seafood consumption scenarios represent the ingestion-weighted average concentration in tissue that results in a certain risk threshold for each scenario. For example, the RBTC for total PCBs for the adult tribal RME seafood consumption scenario based on Tulalip data was 4.2 µg/kg ww at the 1 × 10^{-5} excess cancer risk level. Thus, consumption of 97.5 g/day (adult tribal RME daily ingestion rate based on Tulalip data) of any tissue type with a total PCB concentration of 4.2 µg/kg ww for 70 years would result in a 1 × 10^{-5} excess cancer risk. Consumption of numerous types of seafood, such as crabs, clams, and fish (as specified in the adult tribal RME exposure parameters based on Tulalip data) would also result in a 1 × 10^{-5} excess cancer risk as long as the ingestion-weighted average of the various tissue concentrations consumed was 4.2 µg/kg ww. For the three risk driver chemicals evaluated, the tissue RBTCs for the adult tribal RME scenario based on Tulalip data were lower than all other scenarios for a given risk threshold (Table 8-7).

Table 8-7. Tissue RBTCs estimated from HHRA RME seafood consumption scenarios

CHEMICAL	SEAFOOD CONSUMPTION SCENARIO	UNIT (WW)	TISSUE RBTC[a]			
			EXCESS CANCER RISK			NON-CANCER HAZARD
			1 × 10^{-6}	1 × 10^{-5}	1 × 10^{-4}	HQ = 1
Arsenic (inorganic)	adult tribal RME (Tulalip data)	mg/kg	0.00056	0.0056	0.056	0.25
	child tribal RME (Tulalip data)	mg/kg	0.0030	0.030	0.30	0.12
	adult API RME	mg/kg	0.0019	0.019	0.19	0.37
Total PCBs	adult tribal RME (Tulalip data)	µg/kg	0.42	4.2	42	17
	child tribal RME (Tulalip data)	µg/kg	2.3	23	230	7.8
	adult API RME	µg/kg	1.4	14	140	24
cPAHs[b]	adult tribal RME (Tulalip data)	µg/kg	0.11	1.1	11	na
	child tribal RME (Tulalip data)[c]	µg/kg	0.12	1.2	12	na
	adult API RME	µg/kg	0.39	3.9	39	na

[a] Tissue RBTCs represent ingestion-weighted average concentrations across different seafood categories. Tissue concentrations in individual seafood categories may be higher or lower than the tissue RBTC, but the average concentration in all resident seafood consumed must equal the tissue RBTC in order to result in the specified risk threshold.

[b] cPAH concentrations are expressed in terms of benzo(a)pyrene equivalents.

[c] Because of the potential for increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (2005e), the risk estimate for children for cPAHs is based on dose adjustments across the 0-to-6-year age range of children (see Appendix B, Section B.5.1, for more information).

API – Asian and Pacific Islander
cPAH – carcinogenic polycyclic aromatic hydrocarbon
EPA – US Environmental Protection Agency
HHRA – human health risk assessment
HQ – hazard quotient
na – not applicable
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure
ww – wet weight

Table 8-8. Tissue RBTCs estimated from HHRA non-RME seafood consumption scenarios

CHEMICAL	SEAFOOD CONSUMPTION SCENARIO	UNIT (ww)	TISSUE RBTC[a] EXCESS CANCER RISK 1×10^{-6}	1×10^{-5}	1×10^{-4}	NON-CANCER HAZARD HQ = 1
Arsenic (inorganic)	adult tribal CT (Tulalip data)	mg/kg	0.0085	0.085	0.85	1.6
	child tribal CT (Tulalip data)	mg/kg	0.020	0.20	2.0	0.76
	adult API CT	mg/kg	0.062	0.62	6.2	3.6
	adult tribal (Suquamish)	mg/kg	0.000088	0.00088	0.0088	0.040
	adult one meal per month[b]	mg/kg	0.015	0.15	1.5	2.9
Total PCBs	adult tribal CT (Tulalip data)	µg/kg	6.4	64	640	110
	child tribal CT (Tulalip data)	µg/kg	15	150	1,500	51
	adult API CT	µg/kg	46	460	4,600	240
	adult tribal (Suquamish)	µg/kg	0.066	0.66	6.6	2.6
	adult one meal per month[b]	µg/kg	11	110	1,100	190
cPAHs[c]	adult tribal CT (Tulalip data)	µg/kg	1.7	17	174	na
	child tribal CT (Tulalip data)[d]	µg/kg	0.76	7.6	76	na
	adult API CT	µg/kg	13	130	1,300	na
	adult tribal (Suquamish)	µg/kg	0.018	0.18	1.8	na
	adult one meal per month[b]	µg/kg	3.1	31	310	na

[a] Tissue RBTCs represent ingestion-weighted average concentrations across different seafood categories. Tissue concentrations for individual seafood categories may be higher or lower than the tissue RBTC, but the average concentration in all resident seafood consumed must equal the tissue RBTC in order to match the specified risk threshold.

[b] The adult one-meal-per-month value represents the tissue RBTC that corresponds to consumption of any type of seafood at a rate of one meal per month (7.5 g/day or 0.25 ounces/day).

[c] cPAH concentrations are expressed in terms of benzo(a)pyrene equivalents.

[d] Because of the potential for increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (2005e), the risk estimate for children for cPAHs is based on dose adjustments across the 0-to-6-year age range of children (see Appendix B, Section B.5.1, for more information).

API – Asian and Pacific Islander
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CT – central tendency
EPA – US Environmental Protection Agency
HHRA – human health risk assessment
HQ – hazard quotient
na – not applicable
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
ww – wet weight

For total PCBs, Table 8-9 presents a comparison of tissue RBTCs to empirical LDW and background tissue data for Puget Sound. The tissue RBTC represents an average concentration in seafood associated with a particular diet and a specific consumption rate. Therefore, the tissue RBTC, which are based on a diet of multiple species, and concentrations in a single species are not strictly comparable. Total PCB concentrations in the three species from the LDW were greater than the tissue RBTCs for all risk levels. Total PCB concentrations in tissue from Puget Sound background locations were generally greater than tissue RBTCs at the 1 × 10^{-6} risk level for clams and crabs. For English sole, total PCB concentrations in tissue collected from Puget Sound background locations were generally greater than the tissue RBTC at the 1 × 10^{-5} risk level (Table 8-9).

Table 8-9. Total PCB concentrations in LDW and background tissue compared with tissue RBTCs

	TOTAL PCB CONCENTRATION (µg/kg ww)		
EPC OR RBTC	ENGLISH SOLE	CLAM	DUNGENESS AND SLENDER CRABS
LDW-wide UCL	1,200	600	200
LDW-wide mean	700	140	170
Background (range of mean values)[a]	2.8 – 16	0.12 –- 2.8	0.62 – 1.4
1 x 10^{-4} RBTC (Tulalip RME)	42	42	42
1 x 10^{-5} RBTC (Tulalip RME)	4.2	4.2	4.2
1 x 10^{-6} RBTC (Tulalip RME)	0.42	0.42	0.42

[a] Range of mean total PCB concentrations from various studies and non urban sampling areas in the Puget Sound, as presented in Table 7-19.

EPC – exposure point concentration
LDW – Lower Duwamish Waterway
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
RBTC – risk-based threshold concentration
UCL – upper confidence limit on the mean
ww – wet weight

8.3 SEDIMENT RBTCS FOR HUMAN SEAFOOD CONSUMPTION

Methods for deriving sediment RBTCs for human seafood consumption varied by risk driver. For PCBs, a mechanistic FWM was developed to derive sediment RBTCs (Section 8.3.1). However, such a model is not appropriate for arsenic and cPAHs, so regression models were evaluated for those risk drivers (Sections 8.3.2 and 8.3.3, respectively).

Dioxins and furans were identified as a risk driver for human seafood consumption based on an assumption of unacceptable risk, although no tissue data were available when the risk assessments were conducted. Because estimating sediment RBTCs for the seafood consumption pathway requires modeling the relationship between empirical sediment and tissue concentrations, sediment RBTCs for dioxins and furans could not be estimated for the seafood consumption pathway (sediment RBTCs for dioxins and furans based on direct contact were derived in Section 8.1). Remedial decision-making for dioxins and furans by EPA and Ecology will be based on appropriate guidance and regulations, including those specifically related to the role of background concentrations in the CERCLA and MTCA processes (EPA 2002a; Ecology 2001a).

8.3.1 PCB sediment RBTCs for human seafood consumption

8.3.1.1 Methods for calculating total PCB sediment RBTCs for human seafood consumption

For PCBs, the FWM was used to estimate sediment RBTCs. Information regarding the specific input parameters, calibration, and performance of the FWM is presented in Appendix D (Section D.9), which describes how the FWM was used to derive sediment RBTCs. A best-fit RBTC was estimated using the FWM parameter set that most closely fit the empirical PCB tissue data for all species combined. In addition, a range of sediment RBTCs for total PCBs was estimated for each seafood exposure scenario. The RBTC range represents the range of estimates generated by the FWM while still meeting the FWM performance criterion, which required that the estimated concentrations in tissues be within a factor of 2 of the empirical data for all of the target species. A full discussion of the sources of uncertainty in the FWM is presented in Appendix D, Section D.6.

8.3.1.2 Results of total PCB sediment RBTC calculations for human seafood consumption

Sediment RBTCs were calculated to represent the total PCB concentrations in sediment that correspond to a range of excess cancer risk thresholds (1×10^{-4}, 1×10^{-5}, and 1×10^{-6}) for each of the HHRA scenarios evaluated (Tables 8-10 and 8-11 for the RME and non-RME scenarios, respectively). Sediment RBTCs were estimated to be < 1 µg/kg dw at the 1×10^{-5} or 1×10^{-6} target risk levels for the adult and child tribal RME scenarios and for the adult API RME scenario because the water concentrations assumed in the model (0.0006 to 0.0012 µg/L; see Appendix D, Section D.9) resulted in estimated tissue concentrations greater than the tissue RBTCs in Table 8-7, even in the absence of any contribution from sediment.

Table 8-10. Sediment RBTCs for total PCBs based on excess cancer risks in HHRA RME seafood consumption scenarios

SCENARIO	RISK LEVEL	SEDIMENT RBTC (µg/kg dw)[a]		
		LOWER BOUND[b]	BEST FIT	UPPER BOUND[b]
Adult tribal RME (Tulalip data)	1×10^{-4}	< 1	7.3	25
Child tribal RME (Tulalip data)		109	185	301
Adult API RME		67	100	167
Adult tribal RME (Tulalip data)	1×10^{-5}	< 1	< 1	< 1
Child tribal RME (Tulalip data)		< 1	< 1	< 1
Adult API RME		< 1	< 1	< 1
Adult tribal RME (Tulalip data)	1×10^{-6}	< 1	< 1	< 1
Child tribal RME (Tulalip data)		< 1	< 1	< 1
Adult API RME		< 1	< 1	< 1

[a] For RBTCs presented as < 1 µg/kg dw, a sediment RBTC could not be calculated; even if the total PCB concentration in sediment was set equal to 0 µg/kg dw, FWM-estimated total PCB concentrations in tissue would be greater than the tissue RBTC for the applicable risk level because of the contribution of PCBs from water alone.

[b] Upper and lower bounds were calculated using estimates from parameter sets that met the FWM performance criterion (i.e., empirical data were within a factor of 2 of FWM-estimated concentrations for all species) (see Section D.9).

API – Asian and Pacific Islander
dw – dry weight
FWM – food web model
HHRA – human health risk assessment
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure

Table 8-11. Sediment RBTCs for total PCBs based on excess cancer risks in the non-RME HHRA seafood consumption scenarios

SCENARIO	RISK LEVEL	SEDIMENT RBTC (µg/kg dw)[a, b]		
		LOWER BOUND[c]	BEST FIT	UPPER BOUND[c]
Adult tribal CT (Tulalip data)	1×10^{-4}	370	> 380	> 380
Child tribal CT (Tulalip data)		> 380	> 380	> 380
Adult API CT		> 380	> 380	> 380
Adult tribal (Suquamish data)		< 1	< 1	< 1
Adult one meal per month of benthic fish		220	320	> 380
Adult one meal per month of pelagic fish		170	250	> 380
Adult one meal per month of crabs		> 380	> 380	> 380
Adult one meal per month of clams		> 380	> 380	> 380
Adult tribal CT (Tulalip data)	1×10^{-5}	13	29	62
Child tribal CT (Tulalip data)		68	100	181
Adult API CT		260	> 380	> 380
Adult tribal (Suquamish data)		< 1	< 1	< 1
Adult one meal per month of benthic fish		9	13	23
Adult one meal per month of pelagic fish		< 1	< 1	< 1
Adult one meal per month of crabs		100	220	> 380
Adult one meal per month of clams		270	350	> 380
Adult tribal CT (Tulalip data)	1×10^{-6}	< 1	< 1	< 1
Child tribal CT (Tulalip data)		< 1	< 1	< 1
Adult API CT		0.2	10	28
Adult tribal (Suquamish data)		< 1	< 1	< 1
Adult one meal per month of benthic fish		< 1	< 1	< 1
Adult one meal per month of pelagic fish		< 1	< 1	< 1
Adult one meal per month of crabs		< 1	< 1	< 1
Adult one meal per month of clams		< 1	2	8

[a] For RBTCs presented as < 1 µg/kg dw, a sediment RBTC could not be calculated; even if the total PCB concentration in sediment was set equal to 0 µg/kg dw, FWM-estimated total PCB concentrations in tissue

would be greater than the tissue RBTC for the applicable risk level because of the contribution of PCBs from water alone.

[b] A sediment RBTC of > 380 µg/kg dw indicates that even under current conditions in the LDW, excess cancer risks are estimated to be less than the target risk level.

[c] Upper and lower bounds were calculated using estimates from parameter sets that met the FWM performance criterion (i.e., empirical data were within a factor of 2 of FWM-estimated concentrations for all species) (see Section D.9).

API – Asian and Pacific Islander
CT – central tendency
dw – dry weight
FWM – food web model
HHRA – human health risk assessment
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure

At the 1 × 10^{-4} target risk level, sediment RBTCs were 7.3 and 185 µg/kg dw for the adult and child tribal RME scenarios, respectively, and ranged from < 1 to 25 µg/kg dw for the adult tribal RME scenario and from 109 to 301 µg/kg dw for the child tribal RME scenario (Table 8-10). Sediment RBTCs for the non-RME (informational) scenarios ranged from < 1 µg/kg dw for the adult tribal scenario based on Suquamish data to > 380 µg/kg dw for a number of scenarios at the 1 × 10^{-4} target risk level (Table 8-11). A sediment RBTC of > 380 µg/kg dw indicates that even under current conditions in the LDW,[134] excess cancer risks are estimated to be less than the target risk level. For RBTCs presented as < 1 µg/kg dw, a sediment RBTC could not be calculated; even if the total PCB concentration in sediment was set equal to 0 µg/kg dw, FWM-estimated total PCB concentrations in tissue would be greater than the tissue RBTC for the applicable risk level because of the contribution of PCBs from water alone. This issue arises even at total PCB concentrations in water (0.0003 µg/L) that are similar to the total PCB concentrations in the Green River, upstream of the site (see Table 7-10). Further explanation is provided in Appendix D, Section D.9.

Sediment RBTCs for total PCBs corresponding to non-cancer hazards for RME scenarios with a target HQ of 1 were also estimated; all were < 1 µg/kg dw (Table 8-12). Best-fit RBTCs for the non-RME scenarios ranged from < 1 to > 380 µg/kg dw (Table 8-13).

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[134] A sediment SWAC of 380 µg/kg dw was used in the FWM because it reflected the most current sediment interpolation at the time of model calibration. A subsequent interpolation (with a new IDW parameterization and the inclusion of Round 3 sediment data) resulted in a SWAC of 350 µg/kg dw for the LDW. Appendix D, Section D.6.2.3, presents an analysis of the FWM sensitivity to the SWAC; the average SPAF for all target species was 1.1 regardless of whether the SWAC was 350 or 380 µg/kg dw.

Table 8-12. Sediment RBTCs for total PCBs for the RME HHRA seafood consumption scenarios based on non-cancer hazard quotient of 1

SCENARIO	SEDIMENT RBTC (µg/kg dw)[a]		
	LOWER BOUND[b]	BEST FIT	UPPER BOUND[b]
Adult tribal RME (Tulalip data)	< 1	< 1	< 1
Child tribal RME (Tulalip data)	< 1	< 1	< 1
Adult API RME	< 1	< 1	< 1

[a] Sediment RBTCs were presented as < 1 µg/kg dw because they could not be calculated; even if the total PCB concentration in sediment was set equal to 0 µg/kg dw, FWM-estimated total PCB concentrations in tissue would be greater than the tissue RBTC for the applicable risk level because of the contribution of PCBs from water alone.

[b] Upper and lower bounds were calculated using estimates from parameter sets that met the FWM performance criterion (i.e., empirical data were within a factor of 2 of FWM-estimated concentrations for all species) (see Section D.9).

API – Asian and Pacific Islander
dw – dry weight
FWM – food web model
HHRA – human health risk assessment
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure

Table 8-13. Sediment RBTCs for total PCBs for the non-RME HHRA seafood consumption scenarios based on non-cancer hazard quotient of 1

SCENARIO	SEDIMENT RBTC (µg/kg dw)[a, b]		
	LOWER BOUND[c]	BEST FIT	UPPER BOUND[c]
Adult tribal CT (Tulalip data)	43	76.9	116
Child tribal CT (Tulalip data)	3	15.1	39
Adult API CT	117	193	304
Adult tribal (Suquamish data)	< 1	< 1	< 1
Adult one meal per month of benthic fish	27	40	63
Adult one meal per month of pelagic fish	5	20	27
Adult one meal per month of crabs	211	> 380	> 380
Adult one meal per month of clams	> 380	> 380	> 380

[a] For RBTCs presented as < 1 µg/kg dw, a sediment RBTC could not be calculated; even if the total PCB concentration in sediment was set equal to 0 µg/kg dw, FWM-estimated total PCB concentrations in tissue would be greater than the tissue RBTC for the applicable risk level because of the contribution of PCBs from water alone.

[b] A sediment RBTC of > 380 µg/kg dw indicates that even under current conditions in the LDW, excess cancer risks are estimated to be less than the target risk level.

[c] Upper and lower bounds were calculated using estimates from parameter sets that met the FWM performance criterion (i.e., empirical data were within a factor of 2 of FWM-estimated concentrations for all species) (see Section D.9).

API – Asian and Pacific Islander
CT – central tendency
dw – dry weight
FWM – food web model
HHRA – human health risk assessment
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration

8.3.2 Arsenic sediment RBTCs for human seafood consumption

As discussed in the HHRA (Appendix B), 96% of inorganic arsenic exposure, and hence, 96% of arsenic risk associated with seafood consumption is from consumption of clams. Consequently, the relationship between inorganic arsenic concentrations in clams and sediment is of considerable importance. Tissue chemistry data were collected in 2004 for the Eastern soft-shell clam (*Mya arenaria*). This species was selected for sampling because it is the most abundant clam in the LDW within the size range of interest from a human seafood consumption perspective. Clams in the size range appropriate for human consumption were defined as being over 2 cm long, measured at the widest diameter across the shell (anterior to posterior) (Harbo 2001), or heavier than 16 g with the shell on.[135] Also, *M. arenaria* can tolerate lower salinity environments better than can other native species, such as butter clams (*Saxidomus giganteus*) and native littleneck clams (*Protothaca staminea*). *M. arenaria* are filter feeders, like many other commercially important Puget Sound clam species, such as butter clams and Manila clams (*Tapes philippinarum*). *M. arenaria* can process large quantities of water (4 to 6 L/hr) while feeding on suspended particulates in the water column (Brown et al. 1994).

Given the feeding behavior of *M. arenaria*, water and suspended material in the water are likely to be important sources of bioaccumulated arsenic (Abraham and Dillon 1986). Arsenic was analyzed in hundreds of surface water samples in Elliott Bay, both East and West Waterways, and in the LDW as part of the King County WQA conducted from 1996 to 1997 (King County 1999e). Arsenic was also analyzed in numerous water samples collected in the Green/Duwamish River upstream of the site by King County (2002b, 2007f) (see Section 7.2.4.2 for more details). Mean arsenic concentrations in water at RM 1.0 and RM 2.0 were higher than those at RM 5.0 (Table 8-14). The downstream arsenic concentrations were similar to arsenic concentrations in Elliott Bay; arsenic concentrations in the water at RM 5.0 were similar to those in water collected from RM 10.5 (see Section 7.2.4.2 for more details).

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[135] *M. arenaria* clams were the largest clams collected in the LDW (mean total wet weight of 15.2 g in the 2003 clam survey) (Windward 2004c). This species also had the widest range in weight (0.3 to 54 g). During the 2003 clam survey, 20% of the clams collected in the LDW were *M. arenaria.* Over half the clams collected in the LDW during the 2003 clam survey were identified as *Macoma balthica* (60%), and 18% of the clams were *Macoma nasuta*. Other less common *Macoma* species included *Macoma inquinata* and *Macoma secta* (both < 1%). *M. balthica* were the smallest clams collected on all beaches, with a mean total wet weight of 0.7 g (range of < 0.1 to 3.4 g); whereas *M. nasuta* were on average larger in size (mean total wet weight of 5.2 g, range of 0.3 to 16 g) but too small for human consumption. *M. inquinata* and *M. secta* were similar in size to *M. balthica*.

Table 8-14. Arsenic concentrations in water collected from the LDW and surrounding water bodies

Location	Dissolved Arsenic (µg/L)				Total Arsenic (µg/L)			
	Number of Detects/ Number of Samples	Range	Mean	Median	Number of Detects/ Number of Samples	Range	Mean	Median
Elliott Bay								
North end of waterfront	32/32	0.79 – 1.5	1.1	1.1	79/79	0.27 – 1.5	1.1	1.1
Outer Elliott Bay	22/22	0.96 – 1.5	1.2	1.2	52/52	0.19 –1.5	1.2	1.2
South end of waterfront	72/72	0.68 – 1.5	1.2	1.2	167/167	0.16 – 1.5	1.1	1.2
East and West Waterway								
West Waterway	72/72	0.30 – 1.5	0.98	1.1	165/165	0.30 – 1.5	0.97	1.0
East Waterway	71/71	0.51 – 1.4	1.1	1.2	168/168	0.29 – 1.5	1.1	1.2
LDW								
RM 1.0	72/72	0.24 – 1.5	0.86	0.97	167/167	0.29 – 1.5	0.93	0.89
RM 2.0	72/72	0.22 – 1.4	0.81	0.97	167/167	0.28 – 1.6	0.88	0.77
RM 5.0	24/24	0.18 – 0.46	0.32	0.33	56/56	0.18 – 0.87	0.49	0.48
Upstream of LDW[a]								
Green River at Fort Dent (~RM 10.5 to RM 11.0)	80/104	0.25 – 0.85	0.52	0.54	101/104	0.25 – 4.2	0.87	0.76

Source: King County (1999e)

[a] Upstream water data were collected across a range of conditions, including storms (King County 2002b, 2007f) (see Section 7.2.4.2 for more details).

LDW – Lower Duwamish Waterway

RM – river mile

There is no clear relationship between the arsenic concentration in water and the arsenic concentration in clam tissue samples collected throughout the LDW (i.e., total arsenic concentrations in clams were similar throughout the waterway except for samples collected in the embayment between RM 1.4 and RM 1.5 and in Slip 4, where concentrations were higher in clam tissue). Thus, although filter-feeding clams are likely exposed to arsenic in water (Abraham and Dillon 1986), it is not clear whether the arsenic concentration in water contributes significantly to differences in the arsenic concentration in LDW clam tissues.

Co-located sediment samples were collected in 2004 with the *M. arenaria* clam samples (Windward 2005a); these sediment samples were analyzed for a number of analytes, including total arsenic, grain size, and TOC. Relationships between arsenic concentrations in clam tissues and these co-located sediment samples were investigated; however, studies in the literature suggest that the relationship between the arsenic concentrations in co-located sediment samples and *M. arenaria* tissues is probably tenuous. For example, Foster et al. (1987) conducted an experiment using sediment from Chesapeake Bay to evaluate the bioaccumulation of sediment-sorbed chemicals in a filter-feeding clam (*M. arenaria*) and a deposit-feeding clam (*Macoma balthica*). The *Macoma* clams bioaccumulated the chemicals (naphthalene, diphenyl ether, di-n-octyl phthalate, 4,4′-DDT, and chrysene) at measurable rates over the 12-day exposure period, whereas *Mya* did not show bioaccumulation of any of the sediment chemicals (Foster et al. 1987). These data suggest that a significant relationship between chemical concentrations in bedded sediment and *M. arenaria* in the LDW may be unlikely.

Nevertheless, the relationship between arsenic in clams and sediment collected from the LDW was evaluated using a regression model. Empirical regressions are commonly used to describe the relationship between chemical concentrations in sediment and biota at contaminated sites. Regression models are considered superior to simple uptake factors (e.g., biota-sediment accumulation factors, bioaccumulation factors, bioconcentration factors) because of their ability to address thresholds and other non-linear properties associated with bioaccumulation (Suter 2007).

Because *M. arenaria* are filter feeders, the appropriate areal extent of the source of suspended particulates is likely to be greater than the single co-located composite sediment sample that was collected with the clams. Thus, the regression relationship between the concentrations of inorganic arsenic in clam tissue and total arsenic in sediment was evaluated at three spatial scales. A linear regression model was developed using log -10 transformed concentrations of individual sediment samples as the independent variable and co-located tissue concentrations as the dependent variable. The second regression evaluated the relationship between SWACs of arsenic in sediment from the intertidal areas where clams were collected and the clam tissue concentrations from these areas. The third regression evaluated the SWACs for larger areas surrounding the intertidal areas (e.g., buffer areas, defined as approximately two times the intertidal area itself). Clam tissue sampling locations are shown on Map 8-1 along with the co-located sediment exposure areas over which SWACs were calculated. Only data from exposure areas with detected arsenic concentrations in tissue were used in the analysis (i.e., only 8 of 15 areas were included).

For all of the regression analyses, inorganic arsenic tissue concentrations, rather than total arsenic tissue concentrations, were used to evaluate the relationship between arsenic concentrations in clam tissue and total arsenic concentrations in sediment because the excess risk estimates for seafood consumption were based on the toxicity of inorganic arsenic. Table 8-15 presents the results of the regression analyses for the various tissue/sediment combinations for clams collected in 2004. Additional regression results (including paired data, regression results, and residual tests) are presented in Appendix E, Section E-5. Arsenic was also analyzed in clams collected in 2007. The 2007 data are discussed further in Section 8.3.2.5.

Table 8-15. Results of arsenic regression analyses

SEDIMENT DATASET	n[a]	REGRESSION EQUATION	R^2 VALUE	SIGNIFICANCE OF REGRESSION
Co-located sediment (2004 clam data, including highest value)	8	tissue = [2.6 x (Log(sediment))] - 0.86	0.76	p = 0.005
Co-located sediment (2004 clam data, excluding highest value)	7	tissue = [6.5 x (Log(sediment))] - 3.5	0.71	p = 0.02
Intertidal area SWAC[b]	8	tissue = [2.8 x (Log(sediment))] - 1.4	0.68	p = 0.01
Intertidal area plus buffer SWAC[b]	8	tissue = [3.8 x (Log(sediment))] - 2.75	0.61	p = 0.02

[a] All paired data in which arsenic was detected in both clams and sediment were included.

[b] The SWAC was calculated using the 2008 IDW layer for arsenic.

IDW – inverse distance weighting

n – number of data pairs

SWAC – spatially weighted average concentration

Overall, these data explorations illustrate the uncertainty in the relationship between inorganic arsenic concentrations in clams and total arsenic concentrations in various sediment exposure areas. The regression statistics for the larger intertidal areas were similar to those for the co-located samples. Because expanding the clam exposure areas did not improve the regression relationship, the co-located sediment and tissue data were selected for use in the detailed evaluation of an arsenic regression model.

When developing a regression model, certain assumptions must be met for summary statistics (regression coefficients, R^2) and significance results (p values) to be interpreted exactly. These assumptions include: 1) the relationship between the two variables is linear (or can be made linear using transformations), 2) the residuals around the model (the distribution of the empirical Y data around the regression line) are normally distributed, and 3) the variance of the residuals around the model is homogeneous across the range of predicted values. The assumption requiring normality of the residuals is often relaxed as long as the variance of the residuals is homogeneous. Quantitative tests may also be performed to determine the effect of individual data pairs on the regression coefficients (e.g., “Cook's Distance”). When any of the assumptions of a regression are not met, the validity of a linear model should be investigated and inference about regression results should be drawn with care.

Simple visual diagnostics are often adequate for determining if the assumptions are met, and, if they do not appear to be, whether transformation of one or both variables could help the data to meet test assumptions. The arithmetic arsenic data did not meet the assumption of residual homogeneity of variance, and the distance between the highest data point and the majority of the data was approximately an order of magnitude. Both of these characteristics of the data suggested that a log-transformation would help to linearize the sediment-tissue relationships and equalize variance of the residuals along the line.

After a log-transformation was applied, a significant positive relationship ($R^2 = 0.76$, $p = 0.005$) was found between the eight inorganic arsenic concentrations in LDW clam tissue and the total arsenic concentrations in the eight co-located sediment samples. Figure 8-1 presents the regression model, and Figure 8-2 presents the data on an arithmetic scale.

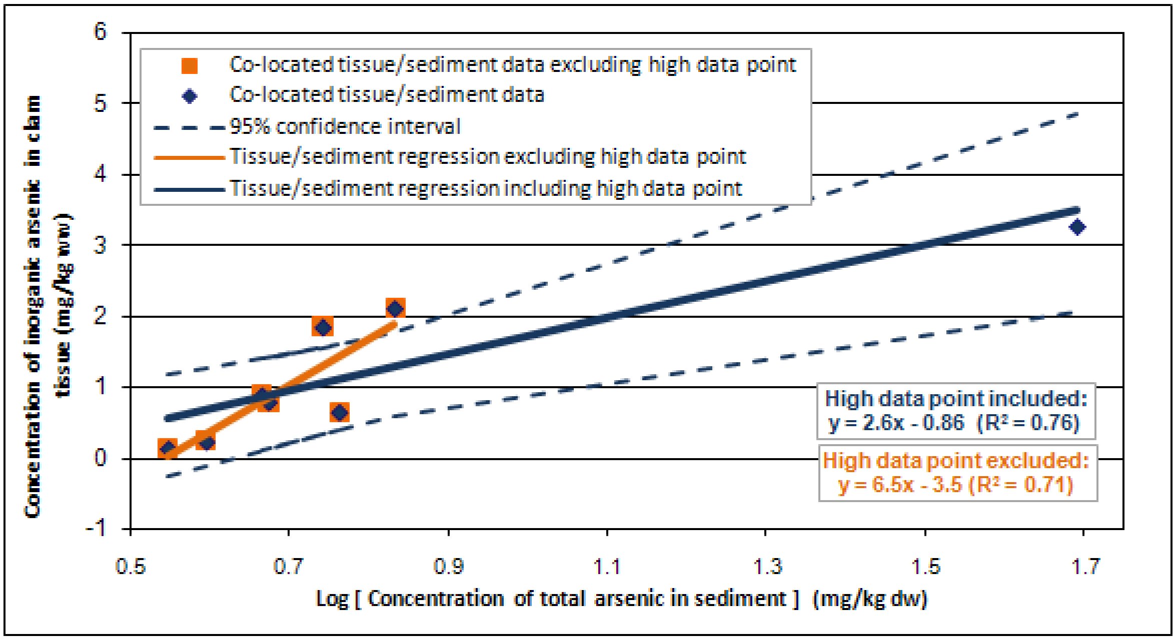

Figure 8-1. **Logarithmic regression of inorganic arsenic concentrations in LDW clam tissue relative to total arsenic concentrations in co-located sediment samples using arithmetic tissue and log[sediment] data**

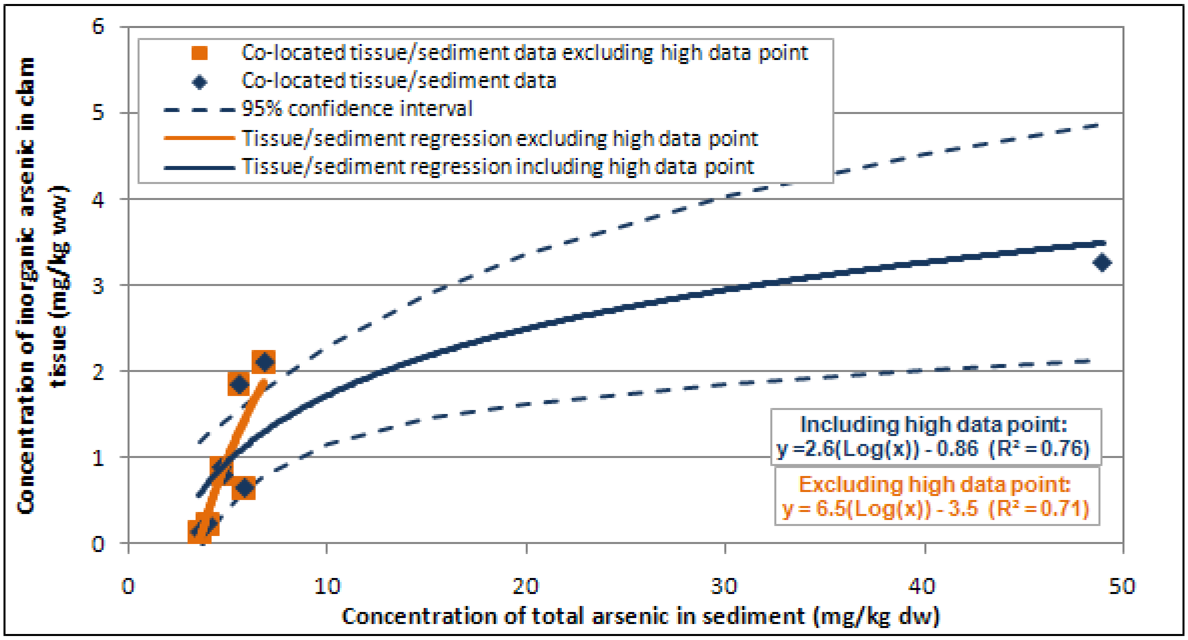

Figure 8-2. Logarithmic regression of inorganic arsenic concentrations in LDW clam tissue relative to total arsenic concentrations in co-located sediment samples presented on arithmetic scale

8.3.2.1 Uncertainties associated with the regression

Although the relationship between the concentration of arsenic in sediment and tissue is significant and positive based on the regression, there are a number of sources of uncertainty that make it imprudent to use a regression model with this dataset to estimate the tissue-sediment relationship for arsenic for the purpose of remedial decision-making. These sources of uncertainty include the:

- Influence of the highest data point on the regression, including the impact of the highest data point on the shape of the regression curve and on the range across which the regression may be applied
- Broad width of the confidence intervals

Each of these sources of uncertainty is described in more detail below.

Uncertainties Associated with the Highest Tissue-Sediment Data Point

The highest sediment concentration is almost an order of magnitude higher than the second highest sediment concentration, but its paired tissue concentration is not similarly higher than the second highest tissue concentration. A regression relationship based on sediment-tissue pairs with sediment concentrations less than or equal to the second-highest sediment concentration would predict a much higher tissue concentration at the highest sediment concentration. The lack of fit of the empirical tissue concentration to its expected value based on the other data creates some uncertainties about the kind of relationship that exists between concentrations in clam tissue and concentrations in sediment. Based on the results of several statistical analyses, the highest tissue-sediment data point had a larger effect than would be expected by chance on: 1) the extent to which the data met the assumptions required to construct a valid linear model, 2) the slope and intercept parameters of the best-fit linear model, and 3) the predictions of tissue concentrations at high sediment concentrations. These findings raise some concerns about which model is correct and how results of a final model should be used.

For example, a primary assumption that must be met for regression significance levels and parameter estimates to be valid is that regression residuals must be normally distributed. One measure of the normality of regression residuals is the correlation between residual values and their expected normal scores. The coefficient of determination between residuals and normal scores for this dataset (0.82) was slightly lower than the threshold (0.90) used by MTCAStat (Ecology 1997) to consider data to be normal, and a Shapiro-Wilks test of the standardized residuals indicated that the residuals may not be normal ($p = 0.032$). Another assumption of a linear regression model is that the model must fit the data along the full range of the data. Several statistical tests indicated that the highest data point may not be part of the same population as the other data points and, consequently, the linear regression model does not fit the entire dataset well. For example, the highest data point's sediment and tissue concentration both qualified as "outliers" with respect to the rest of the values in the dataset.[136] In addition, the sediment and tissue concentrations for the highest data point had a larger than expected influence on the estimated tissue concentration for the highest sediment concentration.[137] Additional statistical analyses indicated that the highest data point also had a greater than expected effect on the estimate of the intercept, the slope, and the regression coefficients.[138]

Although the regression relationship was significant both with and without the highest point, and a high proportion of the variance in tissue concentrations was explained by both regressions ($R^2 > 70\%$ for both regressions; Figure 8-2), the slope of

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[136] The distance between the high sediment concentration and the mean sediment concentration (its leverage value) was 0.82, which is much larger than what would be considered moderate (0.2 to 0.5) or even extreme (> 0.5) leverage values for small datasets (Neter et al. 1990). The high tissue concentration also qualifies as an outlier. Its Studentized deleted residual value of 4.18 is significantly greater than the expected value of 2.6 for this model ($p = 0.009$).

[137] The number of standard deviations between the estimated tissue value using a model with the data point and the estimated tissue concentration using a model without the data point, the DFFit value, was 3.95 compared to an expected DFFit of 1.00.

[138] The standardized difference between the intercepts based on models with and without the data pair, DFbeta0, was 2.67, compared to an expected DFbeta0 of < 1. The standardized difference between the slopes based on models with and without the data point, DFbeta1, was 3.9, compared to an expected DFbeta1 of < 1. Cook's Distance, an overall measure of combined impact of a data point on both regression coefficients, was 25.36, significantly greater than the expected value of 3.5 for this model ($p = 0.001$).

the regression equation was much higher when the highest data point was excluded. In addition, the difference between the slopes of the two regressions had a large effect on the estimated concentration in clam tissue at the highest sediment concentration. If the regression equation using all data was applied, the estimated tissue concentration would have been 3.5 mg/kg ww, while the regression equation excluding the high value would have resulted in an estimated tissue concentration of 7.5 mg/kg ww. This difference in tissue concentrations corresponds to a more than two-fold difference in risk estimates. Although tissue concentrations at lower sediment concentrations might be estimated more accurately using a regression without the highest data point, tissue concentrations at higher sediment concentrations might be greatly overestimated. In addition, excluding the highest data point would limit the range of sediment values across which the regression could be applied (to between 3.5 and 6.8 mg/kg dw) because extrapolation beyond the range of data used to create a regression is not valid.

The distance between the highest data point and the second highest data point and the strong influence of the highest data point on the slope and intercept of the regression equation make it difficult to determine if a linear model is valid or if other explanatory variable(s) (e.g., water concentrations, grain size) are needed in the model to provide more accurate predictions. Inasmuch as all data were collected using the same protocol, there is no substantive reason to exclude the high data point from the dataset; it provides information about areas with higher-than-average arsenic concentrations that may exist in a smaller percentage of the total LDW than do areas with lower concentrations (see Figure 4-21).

Uncertainties Associated with Wide Confidence Intervals

The width of the 95% confidence intervals calculated for the arsenic regression affects the confidence in the estimate of the sediment RBTC from the tissue RBTC. Ideally, the confidence intervals shown on Figure 8-1 could be used to estimate a range of sediment RBTCs at a given tissue RBTC. However, because the sample size is small and the highest sediment concentration is an order of magnitude higher than the mean sediment concentration (sediment concentrations are skewed), the width of the confidence intervals is quite large, especially at higher sediment concentrations. These wide confidence limits, along with the relatively shallow slope of the regression, lead to high uncertainty in inverse predictions of sediment concentrations from risk-based tissue concentrations. For example, given the wide confidence intervals, a sediment concentration of 5 mg/kg dw could be associated with a tissue concentration between 0.4 and 1.6 mg/kg ww with 95% certainty. Assuming an individual consumed only clams at the adult tribal RME seafood consumption rate (97.5 g/day), the excess cancer risks would range from 7 x 10^{-4} to 3 x 10^{-3}.

8.3.2.2 Other efforts to understand the arsenic data

Analytical results for the clam samples collected in 2004 were presented in a data report finalized in June 2005 (Windward 2005a). The proportions of total arsenic consisting of inorganic arsenic in LDW clams were substantially higher than expected based on data from the literature and other sites in Puget Sound (Ecology 2002a). Therefore, additional studies were conducted to further investigate this finding.

Comparison of Results from Different Laboratories

Although the laboratory QA results did not suggest problems with the analyses, a second laboratory was asked to re-analyze a subset of archived clam tissue samples for total and inorganic arsenic to confirm the results. The samples that were analyzed as a result of this request were splits of composite clam tissue samples collected from the LDW in 2004 (LDW-C1-T, LDW-C4-T, LDW-C5-T, and LDW-C6-T), which had been archived frozen. Columbia Analytical Services, Inc. (CAS), analyzed the original samples for total arsenic, and Brooks Rand conducted the analyses for inorganic arsenic. Battelle analyzed the four archived clam sample splits for both total and inorganic arsenic using the same methods used by CAS (EPA 6020) and Brooks Rand (EPA 1632). These four samples were sent to Battelle in May 2005, and the analyses were conducted in June 2005. The differences in sample holding times for these different analyses would not be expected to affect the results.

The concentrations of both total arsenic and inorganic arsenic reported by CAS and Brooks Rand were similar, but consistently higher, than the corresponding concentrations reported by Battelle (Table 8-16). However, the results were not sufficiently different to cast doubt on the original results. Regardless of the laboratory, the proportions of total arsenic consisting of inorganic arsenic were still much higher than those that have typically been identified elsewhere.

Table 8-16. Total and inorganic arsenic concentrations in LDW clam samples analyzed by multiple laboratories

Sample ID	Total Arsenic Concentration (mg/kg ww)			Total Inorganic Arsenic Concentration (mg/kg ww)		
	CAS	Battelle	Percent Difference	Brooks Rand	Battelle	Percent Difference
LDW-C1-T	1.30	0.817[a]	37%	0.132	0.106[a]	20%
LDW-C4-T	5.87	3.29	44%	3.27	2.676	18%
LDW-C5-T	2.28	1.41	38%	0.795	0.512	36%
LDW-C6-T	2.73	2.15	21%	1.85	1.371	26%

[a] Average of two laboratory replicates.

CAS – Columbia Analytical Services, Inc.
ID – identification
LDW – Lower Duwamish Waterway
ww – wet weight

EPA also analyzed splits of three other LDW clam tissue samples at their Manchester laboratory. Total arsenic was analyzed using ICP-MS, which is the same method that was used to analyze LDW samples. Inorganic arsenic and other arsenic species were analyzed using an experimental method that is being developed by EPA, which involves a significantly different extraction. Total arsenic concentrations were very similar between the Brooks Rand and Manchester laboratories (Table 8-17). Inorganic arsenic concentrations were higher for the Brooks Rand results compared with the Manchester results, resulting in a much higher percentage of inorganic arsenic for the clam composite samples collected from other locations. These samples contained various species of clams; clam composite samples from the LDW were composed of *Mya arenaria*, as discussed below.

Table 8-17. Inorganic and total arsenic concentrations in split clam samples

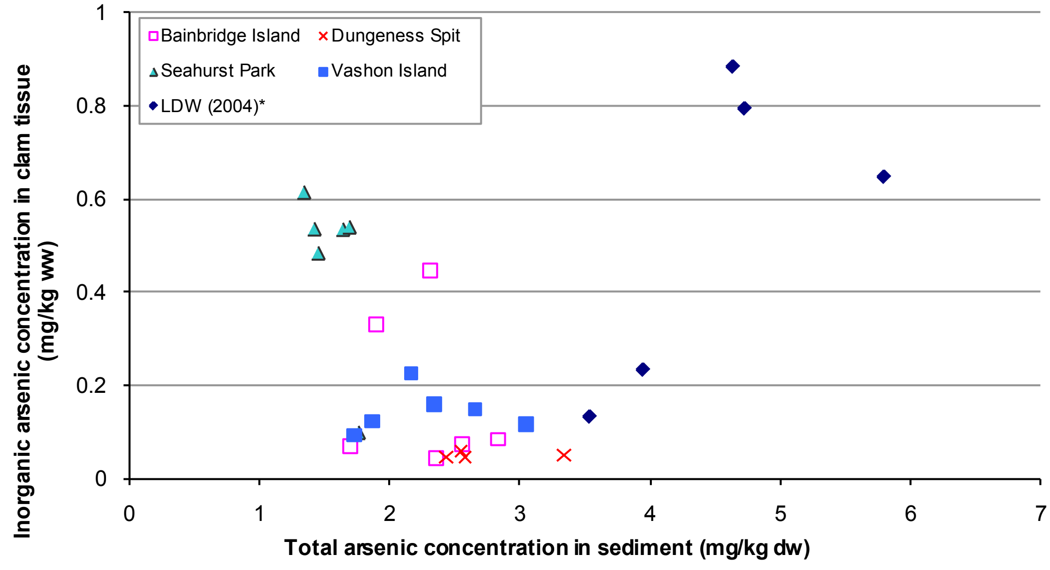
SAMPLE (Location)	MANCHESTER			BROOKS RAND		
	CONCENTRATION (mg/kg ww)		% OF TOTAL ARSENIC THAT IS INORGANIC	CONCENTRATION (mg/kg ww)		% OF TOTAL ARSENIC THAT IS INORGANIC
	INORGANIC ARSENIC	TOTAL ARSENIC		INORGANIC ARSENIC	TOTAL ARSENIC	
SP-C-T4 (Seahurst Park)	0.30	2.02	14.6	0.536	2.04	26.2
LDW-C2-T2 (LDW)	0.31	1.76	17.4	na	1.87	na
BI-C-T6 (Bainbridge Island)	0.21	2.14	9.91	0.331	1.89	17.4

EPA – US Environmental Protection Agency
LDW – Lower Duwamish Waterway
na – not analyzed
ww – wet weight

Analysis of Clams from Other Puget Sound Locations

In 2004, 12 composite clam tissue samples were collected from Seahurst Park (Asarco-influenced background) and Bainbridge Island (outside the area believed to have been affected by the Asarco plume) and analyzed for both total and inorganic arsenic (Windward 2005a) for comparison to the LDW clam data. Because the clam species collected in 2004 from background locations were not the same as the clam species for which arsenic data were available from the LDW, a second background clam sampling event was conducted in 2005. In 2005, 12 composite tissue samples of soft-shell clams (*Mya arenaria,* the same species collected in the LDW) were collected from Vashon Island (Asarco-influenced background) and the Dungeness National Wildlife Refuge (NWR) (outside the area believed to have been affected by the Asarco plume) and analyzed for both total and inorganic arsenic (Windward 2006b).

The relationship between inorganic arsenic in clam tissue and total arsenic in sediment for background *Mya arenaria* samples (collected from Dungeness NWR and Vashon Island) was negative (i.e., inorganic arsenic concentrations in clam tissue decreased with increasing total arsenic concentrations in sediment)(Figure 8-3). Data from species other than *M. arenaria* (collected from Bainbridge Island and Seahurst Park data) also had an inverse relationship between clam tissue and sediment (Figure 8-3), which is the opposite of the relationship noted in LDW clams at higher sediment concentrations (Figures 8-1 and 8-2). The difference in the relationship between inorganic arsenic in LDW clam tissue and total arsenic in sediment compared with clams from other areas of Puget Sound suggests that there may be fundamental differences in the exposure at these locations or in the bioaccumulation dynamics at these lower concentrations. Furthermore, the ratio of inorganic to total arsenic concentrations in clams from these background locations was much lower than that in clams from the LDW, similar to the ratio in clams from other Puget Sound locations. The background *M. arenaria* samples had inorganic arsenic fractions that ranged from 6 to 19%, with a mean of 10%; the other species of clams had a range of 2 to 27%, with a mean of 13% inorganic arsenic. Ratios of total to inorganic arsenic in LDW clams are presented in Table 8-18.



* Only LDW locations with concentrations of inorganic arsenic less than 1.0 mg/kg in clams and concentrations of total arsenic less than 7 mg/kg dw in sediment are shown on this figure.

Figure 8-3. Concentrations of inorganic arsenic in a subset of clams from the LDW compared with concentrations in clams from background areas as a function of total arsenic concentrations in co-located sediment

Table 8-18. Comparison of arsenic concentrations in LDW tissue and sediment samples from the 2004 and 2007 collection efforts

	Non-Depurated Sample Concentrations										Depurated Sample Concentrations			Non-Depurated Sample Concentrations Divided by Depurated Sample Concentrations (2007 data only)	
	2004 Data				2007 Data				2007 Data Divided by 2004 Data		2007 Data				
	Sediment	Clam Tissue			Sediment	Clam Tissue			Tissue	Sediment	Tissue			Tissue	
Station	Total Arsenic (mg/kg dw)	Total Arsenic (mg/kg ww)	Inorganic Arsenic (mg/kg ww)	Inorganic Arsenic (% of Total)	Total Arsenic (mg/kg dw)	Total Arsenic (mg/kg ww)	Inorganic Arsenic (mg/kg ww)	Inorganic Arsenic (% of Total)	Inorganic Arsenic	Total Arsenic	Total Arsenic (mg/kg ww)	Inorganic Arsenic (mg/kg ww)	Inorganic Arsenic (% of Total)	Total Arsenic	Inorganic Arsenic
C1	3.53	1.30	0.132	10%	4.88	2.23	0.690	31%	5.2	1.4	2.35	0.720	31%	0.9	1.0
C2-1	5.79	1.84	0.648	35%	4.53	4.97	2.75	55%	4.2	0.78	3.58	1.13	32%	1.4	2.4
C2-2	3.13	1.88	no data	no data	3.57	4.02	1.73	43%	no data	1.1	2.48	0.920	37%	1.6	1.9
C3-1	4.63	2.01	0.885	44%	5.30	3.91	2.22	57%	2.5	1.1	4.88	1.70	35%	0.8	1.3
C3-2	3.62	4.64	no data	no data	5.27	3.24	1.58	49%	no data	1.5	5.40	2.34	43%	0.6	0.7
C4	49.0	5.87	3.27	56%	172	9.29	6.65	72%	2.0	3.5	12.4	7.60	61%	0.7	0.9
C5	4.72	2.28	0.795	35%	14.1	3.44	1.82	53%	2.3	3.0	4.24	2.17	51%	0.8	0.8
C6	5.52	2.73	1.85	68%	22.4	4.68	4.41	94%	2.4	4.1	7.05	5.72	81%	0.7	0.8
C7-1	6.17	2.82	no data	no data	10.1	6.48	6.40	99%	no data	1.6	11.5	9.30	81%	0.6	0.7
C7-2	6.80	3.86	2.11	55%	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data	no data
C8	10.5	5.44	no data	no data	27.7	5.50	4.10	75%	no data	2.6	8.27	6.25	76%	0.7	0.7
C9	3.94	1.48	0.23	16%	5.62	4.87	2.78	57%	12	1.4	6.76	2.47	37%	0.7	1.1
C10-1	11.9	4.07	no data	no data	37.5	5.13	2.68	52%	no data	3.1	4.67	2.61	56%	1.1	1.0
C10-2	10.8	2.81	no data	no data	7.66	3.56	2.08	58%	no data	0.71	6.27	3.26	52%	0.6	0.6
C11	no data	no data	no data	no data	22.3	2.66	1.37	52%	no data	no data	2.49	1.01	41%	1.1	1.4
C12	no data	no data	no data	no data	67.6	15.2	11.3	74%	no data	no data	19.7	3.28	17%	0.8	3.4
Average	**9.29**	**3.07**	**1.24**	**40%**	**27.4**	**5.28**	**3.50**	**61%**	**4.4**	**2.0**	**6.80**	**3.37**	**49%**	**0.9**	**1.2**

dw – dry weight LDW – Lower Duwamish Waterway ww – wet weight

Collection of Additional Data in the LDW in 2007

In order to further assess inorganic arsenic concentrations in clam tissues, assess the potential effect of depuration, and determine whether or not the high percentage of inorganic arsenic documented in 2004 clam samples was reproducible, 15 additional co-located clam tissue and sediment samples were collected in late summer 2007 and analyzed at Brooks Rand.[139] The concentrations of total arsenic in sediment and inorganic arsenic in clam tissue were higher, on average, in the 2007 data than in the 2004 data (Table 8-18). The percentage of inorganic arsenic remained high.

Half of the clams collected in 2007 were depurated prior to analysis to determine whether depuration could have affected the ratio of inorganic to total arsenic in clam tissue and, if so, if depuration could have affected the relationship between tissue and sediment concentrations. Clams analyzed as part of the 2004 or 2005 RI studies and as part of previous Puget Sound studies (Ecology 2002a) were not depurated. The results of the depuration study are presented in Table 8-18. Although the percentage of inorganic arsenic was generally lower in the depurated samples, no consistent relationship was found between depuration and the ratio of inorganic arsenic to total arsenic in LDW clams.

The 2007 data did not improve the fit of the linear relationship between arsenic concentrations in sediment and clam tissue, as evidenced by lower R^2 values in both the 2007 regression and the regression using 2004 and 2007 data combined (Figure 8-4). The low R^2 value for the combined 2004 and 2007 dataset indicates that a linear regression model does not estimate the relationship between the concentration of inorganic arsenic in tissue and total arsenic in sediment with good precision and thus should not be relied upon for remedial decision-making.[140] Confidence intervals around the regression line provide an indication of the amount of uncertainty in the relationship based on 2004 and 2007 data (Figure 8-5), which is particularly high for extrapolations from a given tissue concentration (e.g., tissue RBTC) to a given sediment concentration.

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[139] In 2007, Softshell clams (*Mya arenaria)* were collected from a total of 15 locations, with one non-depurated and one depurated sample analyzed for each location. *Mya arenaria* weights ranged from 16 to 162 g (average of 53 g) and clam lengths ranged from 45 to 106 mm (average of 74 mm).

[140] The scatter in the data points is particularly pronounced at the low end of the sediment range, where sediment clean-up goals may be set (based on direct–contact sediment RBTCs for arsenic [Table 8-3] and background concentrations presented in Section 7).

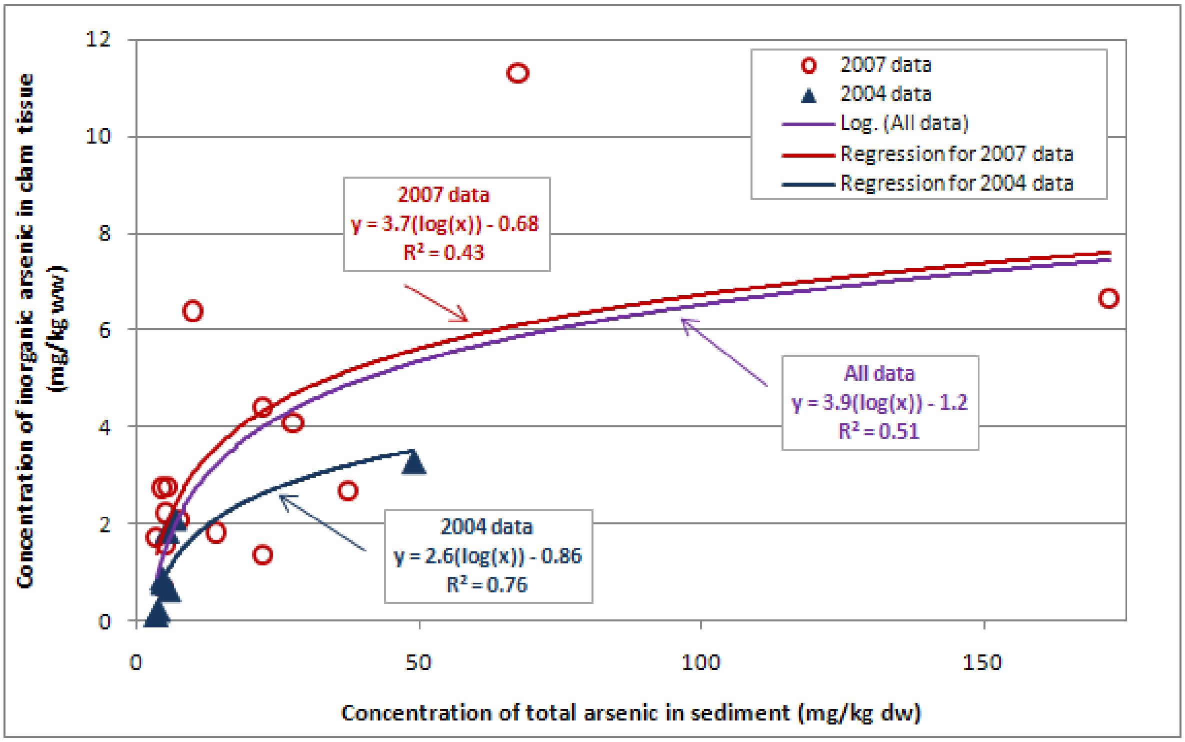


Figure 8-4. Logarithmic regression of inorganic arsenic concentrations in LDW clam tissue relative to total arsenic concentrations in co-located sediment using 2004 and 2007 data

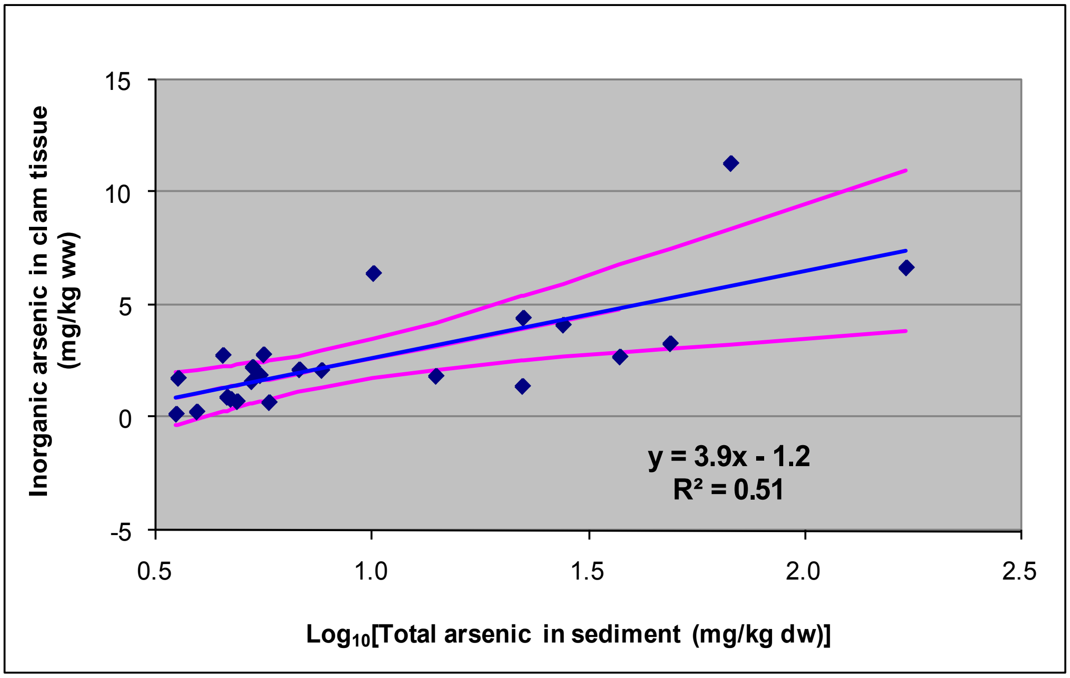


Figure 8-5. Regression relationship for inorganic arsenic in clam tissue and total arsenic in sediment using 2004 and 2007 data

Because the goal of most modeling efforts is to find the simplest model to describe the data, a logarithmic regression was the logical choice for this model application given the premise that the rate of change in the bioaccumulation response generally decreases as the sediment concentration increases (Suter 2007). However, because of the poor logarithmic fit, other statistical models were considered in an attempt to better explain the relationship between inorganic arsenic in tissue and total arsenic in sediment. A log-log regression was computed but did not improve the fit. A quadratic model was not considered appropriate because it would predict decreasing tissue concentrations above an intermediate, but still relatively low, sediment concentration. Other model types that could have been applied to the data (e.g., hockey stick regression) would also have been driven by the high data point. Although the linear model with log transformed sediment was able to explain only 51% of the variance in tissue concentrations, it was the most parsimonious model to describe the data.

8.3.2.3 Development of sediment RBTCs for arsenic and seafood consumption

Based on the information discussed above, LDWG, EPA, and Ecology agree that the clam tissue-to-sediment relationships for arsenic are too uncertain to develop sediment RBTCs. Variables other than sediment concentrations are likely to be important, based on the filter-feeding behavior of *M. arenaria*, and thus any potential effect of sediment remediation on concentrations of inorganic arsenic in clam tissue is highly uncertain. More information, from efforts such as long-term monitoring following sediment remediation and source control, may be needed to determine whether decreases in total arsenic concentrations in sediment affect inorganic arsenic concentrations in clam tissue and, if so, to what extent.

Nevertheless, at the request of EPA, the regression relationship (for the combined 2004 and 2007 dataset) was used to "back-calculate" the sediment concentrations that corresponded to the tissue RBTCs presented in Table 8-7. Because the regressions are not expected to be predictive given the feeding behavior of *M. arenaria*, the development of these sediment concentrations may falsely raise expectations regarding the potential success of sediment cleanup in reducing seafood consumption risks for arsenic. Therefore, this exercise should be viewed as only informational.

Sediment concentrations of total arsenic that corresponded to tissue RBTCs for the RME seafood consumption scenarios (for excess cancer risks between 1 x 10^{-6} and 1 x 10^{-4}, and a non-cancer hazard quotient [HQ] < 1) were all between 2.1 and 2.6 mg/kg dw, even though the tissue RBTCs spanned 3 orders of magnitude and the corresponding confidence intervals spanned an order of magnitude. All of the estimated sediment arsenic concentrations (and even the upper confidence limits) were below the range of 90th percentile arsenic concentrations in upstream surface sediment and Upper Turning Basin subsurface sediments (see Section 7.2). See Appendix E, Section E.7, for more details regarding the back-calculation.

8.3.3 cPAH sediment RBTCs for human seafood consumption

As discussed in the HHRA (Appendix B), 95% of cPAH exposure, and hence, 95% of cPAH risk associated with seafood consumption is from consumption of clams. As discussed in Section 8.3.2, given the feeding behavior of *M. arenaria*, water and suspended particulates in the water are likely to be important sources of bioaccumulated chemicals for this species (Abraham and Dillon 1986). Foster et al. (1987) did not report uptake of PAHs by *M. arenaria* in their study, where the clams "appeared to remain isolated from exposure to contaminated sediment." Nonetheless, the clam tissue/sediment relationship was evaluated to assess the potential to calculate sediment RBTCs for cPAHs associated with seafood consumption in the LDW using site-specific data.

As with arsenic, various regression models were investigated in an attempt to relate site-specific concentrations of cPAHs in composite clam tissue samples and co-located or nearby sediment samples. Multiple spatial scales were evaluated. The performance of the regression model described below was not improved using sediment concentrations averaged over larger spatial areas. The sediment exposure areas evaluated in the cPAH regressions are shown on Map 8-2, and results are presented in Table 8-19.

Table 8-19. Results of cPAH regression analyses

SEDIMENT DATASET	n[a]	REGRESSION EQUATION	R^2 VALUE	SIGNIFICANCE OF REGRESSION
Co-located sediment (including highest value)	14	tissue = [12.8 x (Log(sediment))] - 13.5	0.59	p = 0.001
Co-located sediment (excluding highest value)	13	tissue = [4.89 x (Log(sediment))] + 2.50	0.12	p = 0.02
Intertidal area SWAC	14	tissue = [14.8 x (Log(sediment))] - 23.8	0.49	p = 0.005
Intertidal area plus buffer SWAC	14	tissue = [15.05 x (Log(sediment))] - 24.6	0.36	p = 0.02

[a] All paired data in which cPAHs were detected in both clams and sediment were included.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
n – number of data pairs
SWAC – spatially weighted average concentration

Overall, these data explorations illustrate the uncertainty in the relationship between cPAH concentrations in clams and cPAH concentrations in various sediment exposure areas. Because expanding the clam exposure area and otherwise manipulating the data did not improve the regression relationship, the co-located sediment and tissue data were selected for use in the detailed evaluation of a cPAH regression model.

A logarithmic regression of the co-located data was significant (p = 0.001) and explained 59% of the variance in tissue concentrations when all data were included in the regression. The residuals were normally distributed (fit-R^2 of residuals against expected normal scores = 0.95); however, the variance of the residuals increased with concentration and the pattern of the residuals was not uniform along the range of sediment and tissue concentrations. In addition, a single high data point was responsible for the significance of the regression, as shown in Figure 8-6. The removal of the highest data point reduced the R² value to 0.12. The data and the regression model are presented in Figure 8-6 on a logarithmic scale; Figure 8-7 presents the data and regression on an arithmetic scale. Additional regression results are presented in Appendix E, Section E.5 (including paired data, regression results for co-located sediment-tissue model, and analyses of regression residuals).

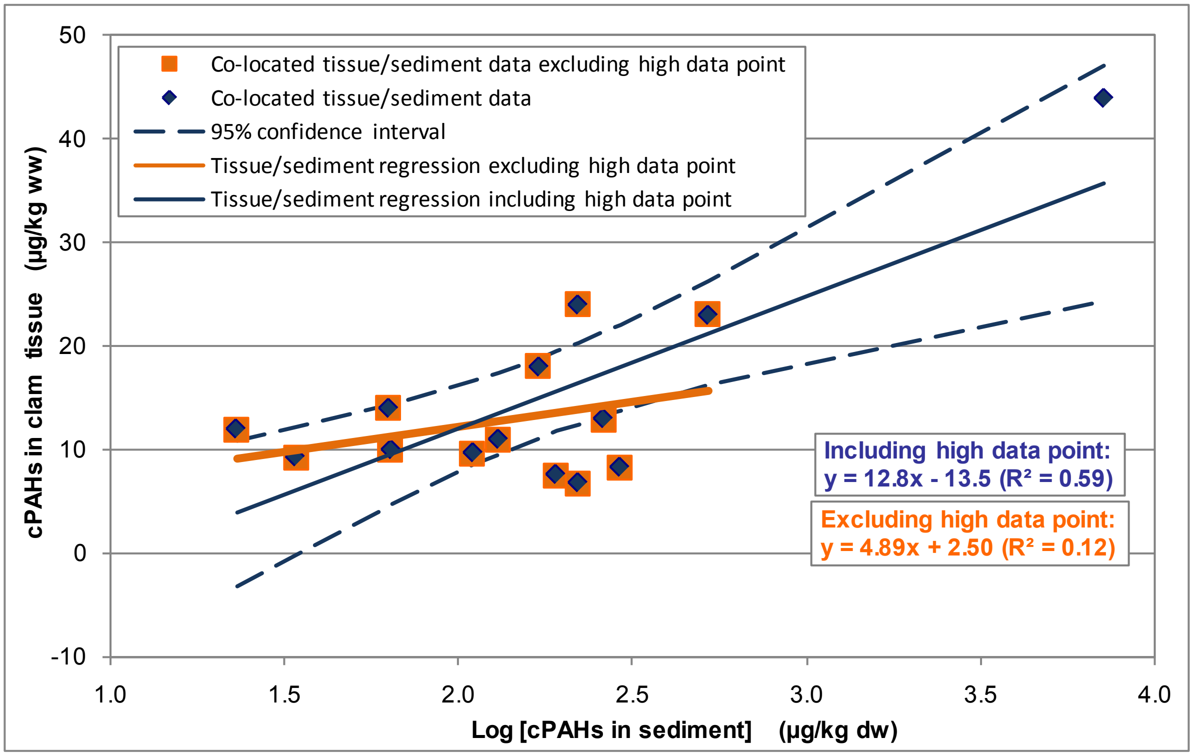


Figure 8-6. Logarithmic regression of cPAH concentrations in LDW clam tissue relative to concentrations in co-located sediment using arithmetic tissue data and log[sediment] data

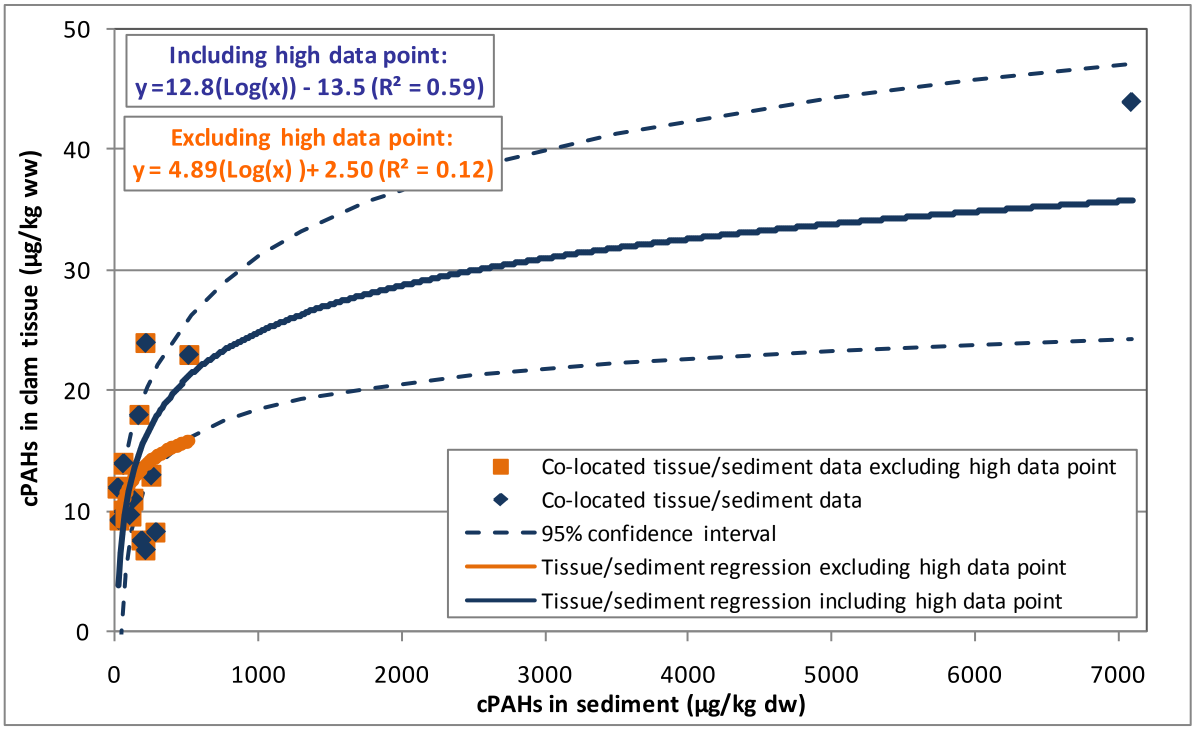


Figure 8-7. Logarithmic regression of cPAH concentrations in LDW clam tissue relative to concentrations in co-located sediment presented on arithmetic scale

As with arsenic, a number of uncertainties make the regression model unsuitable for the purpose of remedial decision-making, including the:

- Impact of the highest data point on the shape and the significance of the regression curve
- Low percent variance explained by the regression (which suggests that other variables may be important)
- Broad width of the confidence intervals

Each of these sources of uncertainty is described in more detail below.

Several outlier diagnostics indicated that the highest tissue-sediment data point had a strong effect on the regression equation. The high data point can be considered an outlier in terms of its sediment concentration (Leverage value = 0.56) and its tissue concentration (Studentized deleted residual = 22.67, $p < 0.0005$). It can also be considered "influential" in its effect on the estimated tissue concentration at its sediment concentration (DFFit = 14.39), its effect on the estimate of the intercept (DFbeta0 = 16.04), and its effect on the estimate of the slope (DFbeta1 = 7.90). The overall influence of the high data point on the regression coefficients was significant at alpha = 0.10 (Cook's Distance = 3.77, p = 0.074). Four other data points also met the criteria defining outliers for both sediment and tissue concentrations. Although these data points did not have the same degree of influence on fitted values or the overall regression equation as the highest data point, they contributed to the irregular pattern in the residuals.

When the highest data point was removed, the regression was no longer significant (R^2 =0.12, p = 0.25). As with arsenic, the highest data point had a large influence on the regression relationship (in this case, it changes not only the slope but also the significance of the relationship), and estimates from one regression would differ greatly from estimates based on the other. Also, removal of the high value would lead to a regression that could only be applied for sediment cPAH concentrations between 23 and 520 µg/kg dw.

In addition to the weaknesses of the cPAH regression model described above, the percent of variance explained by the regression relationship (R^2) is low (particularly when the highest data point is removed). Although the residuals appeared to be normally distributed (Shapiro-Wilks test on standardized residuals p = 0.40), their irregular pattern and increasing magnitude with increasing tissue concentration indicates that other factors besides sediment could be affecting tissue concentrations.

The width of the confidence intervals of the tissue-sediment relationship (Figure 8-6) generates considerable uncertainty in the estimation (i.e., using an inverse prediction) of sediment concentrations from risk-based tissue concentrations, particularly at the low end of the sediment range. Because of the wide confidence intervals, a single sediment concentration may correspond to tissue concentrations that vary by approximately half an order of magnitude (e.g., a sediment concentration of 220 µg/kg dw could correspond to a tissue concentration range of 6.8 to 24 µg/kg ww). Such differences in tissue concentrations correspond to differences in risk estimates of just under an order of magnitude. Assuming an individual consumed only clams at the adult tribal RME seafood consumption rate (97.5 g/day), this range of tissue concentrations would result in excess cancer risks that vary from 3×10^{-5} to 1×10^{-4}. Estimating sediment concentrations that correspond to specific tissue concentrations is equally problematic. For example, according to the regression equation that includes all data, a cPAH concentration of 11 µg/kg ww (the tissue RBTC for adult tribal RME seafood consumption) corresponds to a sediment concentration of 200 µg/kg dw. However, the uncertainty bounds around that concentration range from 40 to > 250 µg/kg dw based on the confidence intervals. The fact that the range of sediment cPAH concentrations that corresponds to a tissue cPAH concentration of 11 µg/kg ww spans nearly an order of magnitude clearly demonstrates the uncertainty in the clam tissue-sediment relationship.

As with the arsenic sediment tissue relationship, more complicated models (e.g., a log-log regression model, hockey stick regression model) were considered. However, for the same reasons discussed in Section 8.3.2.5, a more complicated model was not appropriate for cPAHs based on the data.

Based on the information discussed above, LDWG, EPA, and Ecology agree that the clam tissue-to-sediment relationships for cPAHs are too uncertain to develop RBTCs for sediment. Variables other than sediment concentrations are likely to be important, based on the filter-feeding behavior of *M. arenaria*, and thus any potential effect of sediment remediation on concentrations of cPAHs in clam tissue is highly uncertain; long-term clam tissue monitoring following sediment remediation and source control may be needed to determine whether decreases in cPAH concentrations in sediment affect cPAH concentrations in clam tissue and, if so, to what extent.

Nevertheless, at the request of EPA, the regression relationship was used to "back-calculate" the sediment concentrations that corresponded to the tissue RBTCs presented in Table 8-7. Because the regressions are not expected to be predictive given the feeding behavior of *M. arenaria*, the development of these sediment concentrations may falsely raise expectations regarding the potential success of sediment cleanup in reducing seafood consumption risks for cPAHs. Therefore, this exercise should be viewed as only informational.

cPAH concentrations in sediment (as TEQs) that corresponded to clam tissue RBTCs for the RME seafood consumption scenarios (for excess cancer risks between 1×10^{-6} and 1×10^{-5}) ranged from 12 to 23 µg/kg dw and the corresponding confidence limits spanned 2 orders of magnitude. Sediment concentrations that corresponded to 1×10^{-4} tissue RBTCs for the adult and child tribal RME scenarios were 83 and 100 µg/kg dw, respectively, with corresponding confidence limits that spanned 1 order of magnitude. All of these sediment TEQs are less than or equal to the range of 90th percentile concentrations in upstream surface sediments and Upper Turning Basin subsurface sediments (see Section 7.2). The sediment concentration for the 1×10^{-4} adult Asian and Pacific Islander (API) RME seafood consumption scenario (13,000 µg/kg dw) was much higher than the range of upstream concentrations and the baseline LDW-wide spatially weighted average concentration (SWAC) for cPAHs (380 µg/kg dw). See Appendix E, Section E.7, for more details regarding the back-calculation.

8.4 RBTCS BASED ON RIVER OTTER PREY INGESTION

PCBs were identified as a risk driver chemical for river otter in the ERA (Appendix A). This section presents the derivation of both tissue and sediment RBTCs for this receptor. Both types of RBTCs were calculated because although remediation goals will be sediment-based, tissue RBTCs are relevant for risk communication.

8.4.1 Tissue RBTCs for river otter prey ingestion

Tissue RBTCs were estimated for total PCBs for river otter according to Equation 8-4. To estimate a tissue RBTC corresponding to a HQ of 1, the exposure dose was set equal to the LOAEL that was identified in the ERA (0.26 mg/kg body weight/day). Body weight and food ingestion rate values for river otter were derived from the literature, as described in the ERA (Appendix A, Section A.5.1.2.4). The site use factor was assumed to be 1. PCB dose contributions from water ingestion and incidental sediment ingestion were assumed to be negligible.

$$\text{Tissue RBTC} = \frac{ED \times BW}{SUF \times FIR}$$ **Equation 8-4**

Where:

RBTC = risk-based threshold concentration in prey tissue (mg PCBs/kg food dw)
ED = exposure dose set equal to the LOAEL (mg/kg body weight/day)
BW = body weight (kg)
SUF = site use factor (unitless); fraction of time that river otter spend foraging in the LDW relative to the entire home range
FIR = food ingestion rate (kg food dw/day)

The tissue RBTC for river otter for a LOAEL-based HQ of 1 was 2.7 mg/kg food dw or 540 µg/kg ww (assuming 80% moisture). River otter ingest various fish and shellfish species. The estimated tissue RBTC represents the weighted average concentration of all prey types that could be ingested by river otter.

8.4.2 Sediment RBTCs for river otter prey ingestion

A range of sediment RBTCs for total PCBs was estimated for river otter using the FWM and the same general approach described for the human health consumption scenarios (Section 8.3.1). Section D.9 of Appendix D describes the how the FWM was used to derive sediment RBTCs.

A range of sediment RBTCs for total PCBs was estimated for two dietary scenarios. The range of RBTCs reflects the model performance criterion, which required that the estimated concentrations in tissues were within a factor of 2 of the empirical data for all of the target species. In addition, a best-fit RBTC was estimated using the model parameter set that most closely fit the empirical PCB tissue data for all modeled species combined.

Dietary studies on the river otter indicate that they eat primarily adult fish, although their diet is variable. Because of the possibility that river otter may also consume some juvenile fish, and because juvenile fish were included in the FWM, two dietary scenarios were used to calculate RBTCs: one in which only adult fish were consumed and one in which juvenile fish were consumed along with adults (Table 8-20). Note that the ERA included juvenile chinook salmon as a portion of the river otter's diet as a surrogate for larger salmon. In addition, river otters were assumed to consume crab and bivalves in the ERA. Although there are some differences in the dietary assumptions used in the ERA and those used to derive RBTCs, the RBTCs are considered to be conservative because fish generally had higher total PCB concentrations than crab or bivalves (Section 4.2.3.4).

Table 8-20. Dietary fractions of prey items used in the two river otter dietary scenarios for calculation of RBTCs for total PCBs

RECEPTOR OF CONCERN	DIETARY SCENARIO	FRACTION OF PREY ITEM IN DIET (by weight)					
		CLAM[a]	JUVENILE FISH	CRAB	PACIFIC STAGHORN SCULPIN	SHINER SURFPERCH	ENGLISH SOLE
River otter	including juvenile fish	0.02	0.22	0.1	0.22	0.22	0.22
	excluding juvenile fish	0.02	na	0.1	0.29	0.29	0.29

[a] In the ERA, clams and mussels were assumed to be 1% of the river otter's diet. Because mussels were not modeled in the FWM, clams were assumed to contribute 2% of the river otter's diet for RBTC calculations.

ERA – ecological risk assessment
na – not applicable
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration

Best-fit sediment RBTCs for river otter corresponding to a LOAEL-based HQ of 1 were 159 and 128 µg/kg dw with and without inclusion of juvenile fish in the diet, respectively (Table 8-21). When juvenile fish were excluded from the river otter diet, the sediment RBTCs were slightly lower (ranging from 91 to 217 µg/kg dw) than when juvenile fish were included (ranging from 100 to 250 µg/kg dw). Because total PCB concentrations in juvenile chinook salmon and estimated total PCB concentrations in juvenile fish were lower than total PCB concentrations in Pacific staghorn sculpin, shiner surfperch, and English sole, the overall PCB exposure concentration was lower when juvenile fish were included in the otter diet. Consequently, the corresponding sediment RBTC was higher when juvenile fish were included in the otter diet than when otter were assumed to eat only adult fish.

Table 8-21. Sediment RBTCs for total PCBs for prey consumption by river otter

SCENARIO	RBTCS FOR TOTAL PCBS IN SEDIMENT (µg/kg dw)		
	LOWER BOUND	BEST FIT	UPPER BOUND
Inclusion of juvenile fish in diet	100	159	250
Exclusion of juvenile fish from diet	91	128	217

dw – dry weight
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration

8.5 RBTCS BASED ON BENTHIC INVERTEBRATE EXPOSURE

In the ERA, 41 chemicals were identified as risk drivers for the benthic invertebrate community because they had at least one detected exceedance of the SQS in surface sediment. Because SMS are applicable promulgated standards at the site, sediment RBTCs for these 41 chemicals were set equal to the SQS and the CSL numerical chemical criteria (Table 8-22). Toxicity test data for locations with SMS exceedances must also be considered in the application of these RBTCs because toxicity test information overrides chemical data as part of the SMS. Toxicity test data are presented in Section 4.2.1, and the interpretation of SMS exceedances and site-specific toxicity data are discussed in the ERA (Appendix A). According to the SMS (WAC 173-204-310), locations with all chemical concentrations less than or equal to the SQS are defined as having no acute or chronic adverse effects on biological resources, locations with any chemical concentrations between the SQS and CSL are defined as having minor adverse effects, and locations with any chemical concentration greater than the CSL are defined as having more pronounced adverse effects, pending confirmatory designation via biological testing (WAC 173-204-310-2).

Table 8-22. SMS chemical criteria for risk driver chemicals identified for the benthic invertebrate community

CHEMICAL	UNIT	SQS	CSL
Metals			
Arsenic	mg/kg dw	57	93
Cadmium	mg/kg dw	5.1	6.7
Chromium	mg/kg dw	260	270
Copper	mg/kg dw	390	390
Lead	mg/kg dw	450	530
Mercury	mg/kg dw	0.41	0.59
Silver	mg/kg dw	6.1	6.1
Zinc	mg/kg dw	410	960
PAHs			
2-Methylnaphthalene	mg/kg OC	38	64
Acenaphthene	mg/kg OC	16	57
Anthracene	mg/kg OC	220	1,200
Benzo(a)anthracene	mg/kg OC	110	270
Benzo(a)pyrene	mg/kg OC	99	210
Benzo(g,h,i)perylene	mg/kg OC	31	78
Total benzofluoranthenes	mg/kg OC	230	450
Chrysene	mg/kg OC	110	460
Dibenzo(a,h)anthracene	mg/kg OC	12	33
Dibenzofuran	mg/kg OC	15	58
Fluoranthene	mg/kg OC	160	1,200
Fluorene	mg/kg OC	23	79
Indeno(1,2,3-cd)pyrene	mg/kg OC	34	88
Naphthalene	mg/kg OC	99	170
Phenanthrene	mg/kg OC	100	480
Pyrene	mg/kg OC	1,000	1,400
Total HPAH	mg/kg OC	960	5,300

CHEMICAL	UNIT	SQS	CSL
Total LPAH	mg/kg OC	370	780
Phthalates			
BEHP	mg/kg OC	47	78
Butyl benzyl phthalate	mg/kg OC	4.9	64
Dimethyl phthalate	mg/kg OC	53	53
Other SVOCs			
1,2,4-Trichlorobenzene	mg/kg OC	0.81	1.8
1,2-Dichlorobenzene	mg/kg OC	2.3	2.3
1,4-Dichlorobenzene	mg/kg OC	3.1	9.0
2,4-Dimethylphenol	µg/kg dw	29	29
4-Methylphenol	µg/kg dw	670	670
Benzoic acid	µg/kg dw	650	650
Benzyl alcohol	µg/kg dw	57	73
Hexachlorobenzene	mg/kg OC	0.38	2.3
n-Nitrosodiphenylamine	mg/kg OC	11	11
Pentachlorophenol	µg/kg dw	360	690
Phenol	µg/kg dw	420	1,200
PCBs			
Total PCBs	mg/kg OC	12	65

BEHP – bis(2-ethylhexyl) phthalate
CSL – cleanup screening level
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
OC –organic carbon
PCB – polychlorinated biphenyl
SMS – Washington State Sediment Management Standards
SQS – sediment quality standard
SVOC – semivolatile organic compound

8.6 APPLICATION OF RBTCS

The RBTCs derived in this section are an important consideration in the derivation of PRGs in the FS. Three issues related to application of RBTCs are discussed here: spatial considerations (Section 8.6.1), consideration of cumulative and aggregate risk (Section 8.6.2), and considerations of assumptions regarding current and future use (Section 8.6.3).

8.6.1 Spatial considerations

Appropriate application of RBTCs requires explicit consideration of spatial scale. Some RBTCs are appropriate for application on a point-by-point basis, while most others should be applied only at a larger scale. For example, RBTCs based on the protection of benthic invertebrates (see Section 8.5) should be applied on a point-by-point basis because of the small home ranges of these animals and the regulatory framework described in the SMS. All other RBTCs (Sections 8.1 through 8.4) are most appropriately applied at larger spatial scales. RBTCs for seafood consumption should be applied at an LDW-wide spatial scale because risks were evaluated at this scale.

RBTCs derived for direct sediment contact scenarios should be applied to appropriate exposure areas. The exposure area for the netfishing RME scenario included all of the intertidal and subtidal areas of the LDW. For the beach play RME scenario, eight separate intertidal areas were selected in the HHRA as exposure areas where children are most likely to contact sediment (Map 6-1). These areas could change in the future if, for example, a new park with access to intertidal areas were to be constructed. The exposure area for the RME clamming included all intertidal areas where clamming could occur because such areas are potentially accessible either from a boat or from shore (Map 6-2).

8.6.2 Consideration of cumulative and aggregate excess cancer risk

Following both EPA and Ecology guidance for conducting HHRAs, cumulative excess cancer risk estimates were calculated in the HHRA by summing risk estimates for individual COCs. Cumulative excess cancer risk should also be given consideration during the application of RBTCs in the FS. In practice, cumulative carcinogenic risk can be considered in the application of RBTCs in specific areas by summing the risks for individual chemicals within a given exposure area to ensure that they do not result in an unacceptable cumulative target risk. Cumulative risk for non-cancer hazards can only be evaluated if the chemicals affect the same target organs or have the same endpoint effects.

For example, if the target excess cancer risks for four individual risk drivers were all set to 1×10^{-6} for calculation of individual chemical RBTCs, the cumulative target risk would be no greater than 4×10^{-6}. In this example, if the acceptable target cumulative risk was 1×10^{-5}, then meeting each individual RBTC that was based on a target risk of 1×10^{-6} would result in an acceptable cumulative risk estimate for that area because 4×10^{-6} is less than 1×10^{-5}.

8.6.3 Consideration of assumptions regarding current and future LDW use

The human health RBTCs presented in the RI, particularly those for RME scenarios, are based on exposure assumptions that are intended to be health-protective of both current and future uses of the LDW. There is limited site-specific data available on the amount of resident seafood organisms currently being harvested and consumed from the LDW, and no site-specific data on the degree to which people may currently be exposed via beach play or clamming activities. Consequently, the data used to establish the exposure parameters for these RME scenarios were based on other sources, including tribal seafood consumers and clam diggers from other areas of Puget Sound and beach users from King County lakes. The tribal netfishing scenario, on the other hand, reflects exposure conditions that could occur under current tribal fishing activities within the LDW.

The exposure assumptions for the RME scenarios for tribal seafood consumption and clam digging are likely overestimates of current behavior in the LDW because current use is likely depressed as a result of the fish advisory and local knowledge of contamination. With respect to seafood consumption, WSDOH has issued health advisories for LDW fish and crab, warning the public that any seafood consumption from the LDW is unsafe because of elevated PCB concentrations in these species (WSDOH 2005). WSDOH (Office of Food Safety and Shellfish) and Public Health – Seattle & King County advise against consumption of clams in the LDW because of pathogens from urban sewage releases (WSDOH 2005). Environmental quality, and consequently seafood resources, is expected to improve in the future, but it is not known if future site use will reach the RME exposure levels assumed in the HHRA (Appendix B).

A creel study (in which fishermen were observed and surveyed while fishing) was conducted by King County to assess the amount of fishing activity that occurred in Elliott Bay and the LDW in 1997 (King County 1999c). The study included surveys at eight locations (Table 8-23) within the LDW on 30 different days during the summer of 1997 (between June 22 and August 30), including 10 Saturdays, 10 Sundays, and 10 weekdays. Sites were visited repeatedly between 5 a.m. and 8 p.m. (weekends) and from 5 a.m. to 11 a.m. and 4 p.m. to 10 p.m. (weekdays).

Table 8-23. Locations and number of fishermen identified in the King County creel survey

FISHING LOCATION WITHIN LDW	NUMBER OF INDIVIDUALS OBSERVED FISHING[a]
Spokane Street S Bridge[b]	192
T-105	10
Diagonal Avenue S	2
Duwamish Waterway Park	1
Kellogg Island	0
Muckleshoot 1st Avenue. boat ramp	0
8th Avenue, street end	0
Boeing office building access	0
8th Avenue S and S Portland Street	0

Source: King County (1999c)

[a] Surveys were conducted within the LDW on 30 different days during the summer of 1997 (between June 22 and August 30), including 10 Saturdays, 10 Sundays, and 10 weekdays. Sites were visited repeatedly between 5 a.m. and 8 p.m. (weekends) and from 5 a.m. to 11 a.m. and 4 p.m. to 10 p.m. (weekdays).

[b] The Spokane Street S Bridge is 755 ft north of the LDW study area boundary. Because the home ranges of resident fish and crabs within the LDW have not been explicitly defined, individuals fishing at this location may catch and consume fish and crabs that may have been exposed to LDW-related contamination.

LDW – Lower Duwamish Waterway

T-105 – Terminal 105

Very few individuals were observed fishing in the LDW compared with Elliott Bay (13 in the LDW and over 1,000 in Elliott Bay), and seafood was caught at only one of the eight locations (two crabs and one sole were caught at T-105). Although fishing was documented to be infrequent within the LDW, one of the most popular fishing locations in the survey was the Spokane Street S Bridge, which is 755 ft north of the study area, from which people can fish in the East Waterway. Persons fishing from this bridge may catch and consume seafood that may have been exposed to LDW contamination because fish collected from this location may use LDW habitat to an unknown extent. Although this survey did not evaluate the frequency of fishing from boats, people have been observed using boats to fish for salmon in the LDW.

Future remedial and restoration actions could increase the frequency of fishing and other recreational activities, particularly kayaking and canoeing. Although the LDW is expected to remain an industrial corridor for the foreseeable future, remedial and other actions could result in a reduction in or suspension of some fish advisories, thereby increasing the use of the LDW as a fishing resource. Increased public access and habitat restoration activities may not only expand the areas where people currently access the river, but also increase the frequency with which they do so. Consequently, the potential overestimation of current risks in the exposure scenarios is intended to be protective of future uses as well.

8.7 SUMMARY

RBTCs in sediment were derived for consideration in the development of PRGs in the FS. RBTCs were derived for human direct sediment contact scenarios for arsenic, cPAHs, dioxins and furans, and total PCBs (Table 8-24; Figure 8-8). Tissue RBTCs (Table 8-25) were also derived for risk communication purposes. Sediment RBTCs based on the seafood consumption scenarios were estimated for total PCBs using the FWM. Regression models for relating concentrations of inorganic arsenic and cPAHs in clams to concentrations of arsenic and cPAHs in co-located sediment were evaluated and found to be unsuitable for developing RBTCs. No sediment RBTC based on seafood consumption was estimated for dioxins and furans because dioxin and furan tissue data from the LDW were not available when the risk assessments were conducted to support such a derivation. RBTCs were also estimated for total PCBs in both tissue and sediment for prey consumption by river otter. Sediment RBTCs for benthic invertebrates were set equal to SMS criteria.

Table 8-24. Summary of sediment RBTCs for risk drivers

Risk Driver Chemical and SMS Criteria (if available)	Risk Level	Sediment RBTC for Direct-Contact Scenarios			Sediment RBTC for Human Seafood Consumption Scenarios			Sediment RBTC for Ecological Receptor[a]
		Netfishing RME	Beach Play RME	Tribal Clamming RME	Adult Tribal RME (Tulalip Data)	Child Tribal RME (Tulalip Data)	Adult API RME	River Otter Prey Consumption
Arsenic (mg/kg dw) SQS = 57, CSL = 93	1×10^{-6}	3.7	2.8	1.3	nc[b]	nc[b]	nc[b]	na[c]
	1×10^{-5}	37	28	13	nc[b]	nc[b]	nc[b]	na[c]
	1×10^{-4}	370	280	130	nc[b]	nc[b]	nc[b]	na[c]
	HQ = 1	nc[d]	nc[d]	nc[d]	nc[b]	nc[b]	nc[b]	na[e]
cPAHs (µg/kg dw)	1×10^{-6}	380	90	150	nc[b]	nc[b]	nc[b]	na[c]
	1×10^{-5}	3,800	900	1,500	nc[b]	nc[b]	nc[b]	na[c]
	1×10^{-4}	38,000	9,000	15,000	nc[b]	nc[b]	nc[b]	na[c]
	HQ = 1	nc[d]	nc[d]	nc[d]	nc[b]	nc[b]	nc[b]	na[f]
Dioxin/furan TEQ (ng/kg dw)	1×10^{-6}	37	28	13	ne[g]	ne[g]	ne[g]	ne[g]
	1×10^{-5}	370	280	130	ne[g]	ne[g]	ne[g]	ne[g]
	1×10^{-4}	3,700	2,800	1,300	ne[g]	ne[g]	ne[g]	ne[g]
	HQ = 1	nc[d]	nc[d]	nc[d]	ne[g]	ne[g]	ne[g]	ne[g]
Total PCBs (µg/kg dw) SQS = 12 mg/kg OC, CSL = 65 mg/kg OC	1×10^{-6}	1,300	1,700	500	<1[h]	< 1[h]	<1[h]	na[c]
	1×10^{-5}	13,000	17,000	5,000	< 1[h]	< 1[h]	<1[h]	na[c]
	1×10^{-4}	130,000	170,000	50,000	7.3 (< 1 – 25)[i]	185 (109 – 301)[i]	100 (67 – 167)[i]	na[c]
	HQ = 1	nc[d]	nc[d]	nc[d]	< 1[h]	< 1[h]	< 1[h]	128 – 159[j]

[a] Sediment RBTCs for the remaining 39 risk driver chemicals identified for the benthic invertebrate community were presented in Table 8-22.

[b] Regression models for relating concentrations of arsenic and cPAHs in clams to concentrations of these chemicals in co-located sediment were evaluated and found to be unsuitable for developing RBTCs.

[c] Tissue RBTCs for excess cancer risk were developed only for human health.

[d] Sediment RBTCs for direct-contact scenarios were estimated only for excess cancer risks. Sediment RBTCs were not estimated for non-cancer hazards for direct-contact scenarios because none of the RME scenarios had HQs for an individual chemical > 1 or generated endpoint-specific HIs in > 1 (Appendix B, Section B.5.6).

[e] Tissue RBTCs for arsenic were not developed for otter because arsenic was not a COC for otter.

[f] Tissue RBTCs for cPAHs were not developed for otter because cPAHs were evaluated only for human health; PAHs were evaluated as individual PAH compounds for otter in the ERA and were not a COPC.

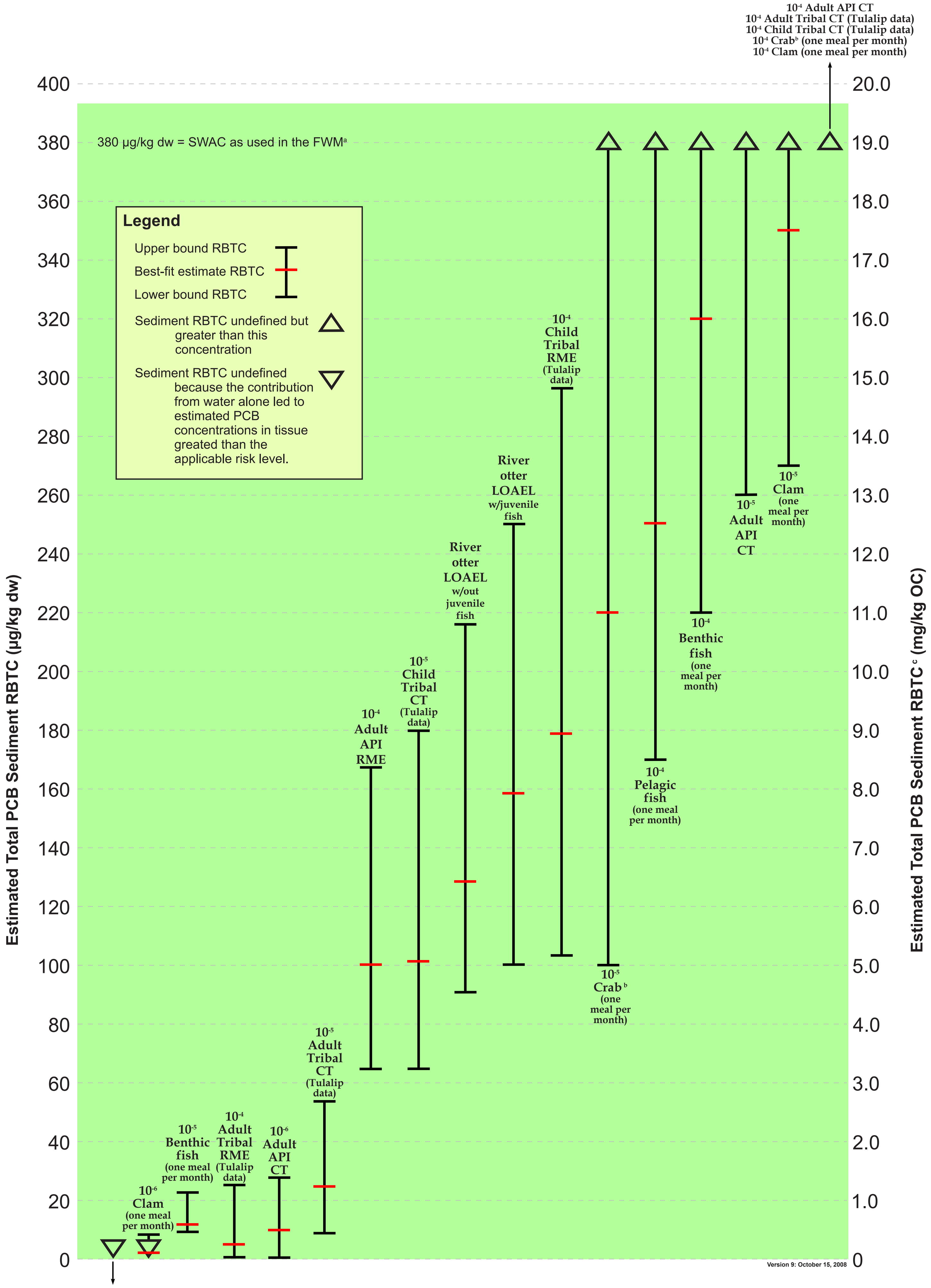
[g] RBTCs, other than those for direct sediment contact human health scenarios, could not be calculated because there were no dioxin and furan tissue data available for the risk assessments.

[h] For RBTCs presented as < 1 µg/kg dw, a sediment RBTC could not be calculated; even if the total PCB concentration in sediment was set equal to 0 µg/kg dw, FWM-estimated total PCB concentrations in tissue would be greater than the tissue RBTC for the applicable risk level because of the contribution of PCBs from water alone.

[i] The RBTC represents best-fit estimate from the FWM and the upper- and lower-bound estimates from FWM parameter sets that met the model performance criterion of estimates for all target species within a factor of 2 of empirical data.

[j] Represents best-fit estimates from the FWM for two different dietary scenarios, as described in Section 8.4.

API – Asian and Pacific Islander
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CSL – cleanup screening level
dw – dry weight
FWM – food web model
HI – hazard index
HQ – hazard quotient
LDW – Lower Duwamish Waterway
na – not applicable
nc – not calculated
ne – not evaluated
ns – not selected (as a risk driver for this receptor)
OC – organic carbon
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure
SMS – Washington State Sediment Management Standards
SQS – sediment quality standards
TEQ – toxic equivalent



Note: Ranges shown incorporate results from the final ERA and final HHRA, results from FWM Calibration 2, and the ranges associated with two different diet assumptions for river otter.

[a] A sediment SWAC of 380 µg/kg dw was used in the FWM because it reflected the most current sediment interpolation at the time of model cal bration. However, subsequent interpolation (with a new IDW parameterization and inclusion of Round 3 sediment data) resulted in a SWAC of 350 µg/kg dw.

[b] Crab one-meal-per-month scenario included consumption of equal portions of slender crabs and Dungeness crabs.

[c] Sediment RBTCs estimated from the FWM were on a dry-weight basis; for the sake of comparison, the dry-weight sediment RBTCs were converted to approximate OC-normalized sediment RBTCs, assuming 2% organic carbon.

Figure 8-8. Summary of sediment RBTCs for total PCBs for selected human seafood consumption and river otter prey consumption scenarios

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Table 8-25. Summary of tissue RBTCs for risk drivers

Risk Driver Chemical	Risk Level	Tissue RBTCs for Human Seafood Consumption Scenarios			Tissue RBTCs for Ecological Receptor
		Adult Tribal RME	Child Tribal RME	Adult API RME	River Otter Prey Consumption
Arsenic (mg/kg ww)	1×10^{-6}	0.00056	0.0030	0.0019	na[a]
	1×10^{-5}	0.0056	0.030	0.019	na[a]
	1×10^{-4}	0.056	0.30	0.19	na[a]
	HQ = 1	0.25	0.120	0.37	na[b]
cPAHs (µg/kg ww)	1×10^{-6}	0.11	0.12	0.39	na[a]
	1×10^{-5}	1.1	1.2	3.9	na[a]
	1×10^{-4}	11	12	39	na[a]
	HQ = 1	na	na	na	na[c]
Total PCBs (µg/kg ww)	1×10^{-6}	0.42	2.3	1.4	na[a]
	1×10^{-5}	4.2	23.0	14	na[a]
	1×10^{-4}	42	230	140	na[a]
	HQ = 1	17	8	24	540

[a] Tissue RBTCs for excess cancer risk were developed only for human health.

[b] Tissue RBTCs for arsenic were not developed for otter because arsenic was not a COC for otter.

[c] Tissue RBTCs for cPAHs were not developed for otter because cPAHs were evaluated only for human health; PAHs were evaluated as individual PAH compounds for otter in the ERA and were not a COPC.

API – Asian and Pacific Islander
cPAH – carcinogenic polycyclic aromatic hydrocarbon
HQ – hazard quotient
na – not applicable
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure
ww – wet weight

Sediment RBTCs for total PCBs under the various human seafood consumption scenarios and for prey consumption by river otters are compared in Figure 8-8. This figure facilitates a visual comparison of the RBTC ranges for the different seafood consumption scenarios. It also shows that across the range of total PCB concentrations of interest in sediment (0 to 380 µg/kg dw), RBTCs range from < 1 µg/kg dw to greater than 380 µg/kg dw, depending upon the seafood consumption scenario and target risk level. The appropriate application of RBTCs to sediment in the LDW must consider the specific exposure areas identified in the HHRA for direct-contact scenarios and for seafood consumption RBTCs.

9 Potential Pathways, Source Identification, and Source Control Efforts

Source control is the process of identifying and preventing or reducing releases of chemicals into the environment. The goal of an effective source control strategy is to minimize the potential for sediment to become recontaminated to levels of concern after remediation has been completed. Since 2001, source identification and control programs in the LDW have been established by regulatory authorities including the City of Seattle, King County, Ecology, and EPA. The goal of these efforts has been to identify potential sources of chemicals to the LDW and to implement cleanup plans and source control strategies.

The LDW and its adjacent upland areas have been used for industrial, commercial, and shipping operations for nearly a century. During this time, chemicals associated wFith those practices have entered the LDW through various pathways. Forty-one chemicals have been detected in sediment from the LDW at concentrations exceeding SMS criteria, including PCBs, PAHs, metals, phthalates, and other SVOCs; dioxins and furans, which do not have SMS criteria, were also elevated in some areas. Many private and agency-led environmental investigations have been conducted both in the LDW and on adjacent upland properties to further characterize the nature and extent of contamination in the LDW and to identify potential sources of chemicals that could continue to enter the LDW.

Ecology, as the lead source control agency, formed the LDW Source Control Work Group (SCWG) in 2002. The SCWG is composed primarily of public agencies responsible for source control, including Ecology, the City of Seattle, the City of Tukwila, King County, the Port of Seattle, and EPA. Public agencies with the authority to regulate source control in the LDW use stormwater programs, permits, hazardous waste management and pollution prevention programs, inspection and maintenance programs, water quality compliance and spill response programs, environmental and pathways assessments, and other regulatory programs in an effort to control sources of contamination. The SCWG is discussed further in Section 9.2.

In 2004, Ecology developed a source control strategy for the LDW (2004b). The basic approach for the LDW source control strategy is to identify and manage sources of sediment contamination in coordination with the LDW-wide RI, as well as more localized, site-specific investigations. The strategy provides the basis for Ecology to identify priority areas through a tiered process and to develop "summary of existing information and data gaps reports" (hereafter referred to as data gaps reports) and detailed area-specific source control action plans (SCAPs) for each priority area. The status of the reports for the various areas is discussed in Sections 9.2.2 and Appendix I.

This section describes the potential migration pathways through which chemicals can enter the LDW (Section 9.1), discusses the overall LDW source control strategy and

SCWG efforts (Section 9.2), identifies the various potential historical and ongoing sources to the LDW (Section 9.3), and discusses the ongoing source identification and control efforts being implemented in the LDW (Section 9.4). The information in this section was compiled using readily available data; historical records and documents; and information from EPA, Ecology, the City of Seattle, the City of Tukwila, the Port of Seattle, The Boeing Company, and King County.

Source control is an ongoing, iterative process that continually produces new information. The summary of source control information provided in this report reflects information that was available at the time of document preparation. Information in this report does not supersede or replace information provided by the SCWG through their source control strategy implementation, including SCAPs, data gaps reports, source control status reports, or other SCWG products. These sources should be referred to for the most up-to-date and complete source control documentation.

The source information presented in this section for the greater LDW drainage basins (Map 9-1) is useful in the development of a CSM for the LDW and will assist in the remedial alternative decision-making process.

9.1 POTENTIAL PATHWAYS

Chemicals released to media such as air, soil, groundwater, and surface water or to impervious surfaces may migrate to the LDW through various potential pathways. As shown in Figure 9-1, potential pathways include atmospheric deposition; surface water inputs; direct discharge into the LDW (e.g., CSOs, storm drains); chemical spills and/or leaks to the ground, surface water, or directly into the LDW; groundwater migration; and bank erosion. This section briefly highlights the potential pathways to the LDW. In-depth information is provided in Sections 9.3 and 9.4.

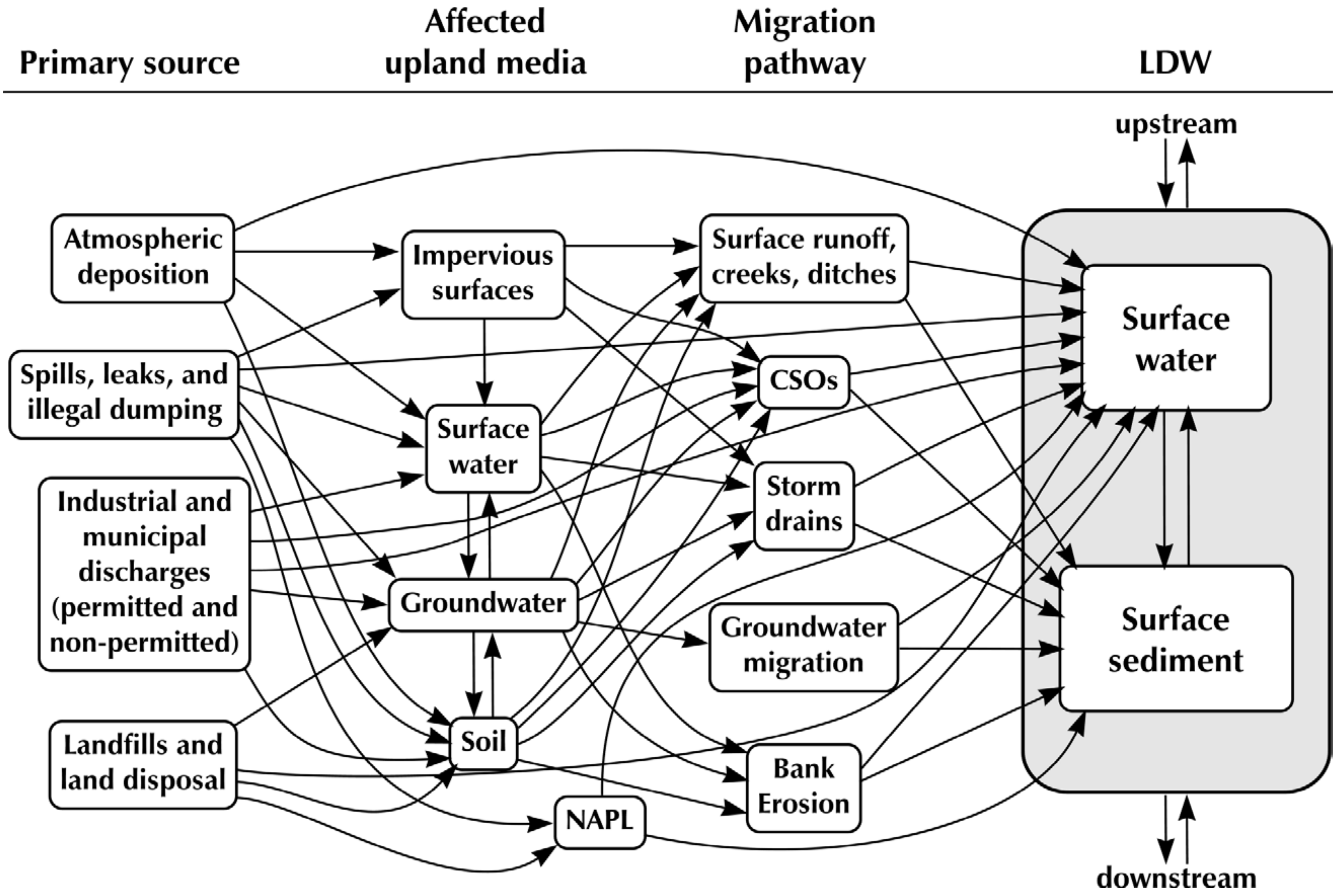

Figure 9-1. Conceptual model of chemical sources and potential pathways to the LDW

9.1.1 Atmospheric deposition

Chemicals are emitted to the air from both point and non-point sources. Point sources include various industrial facilities (EPA 2001). Non-point sources include mobile emissions from motor vehicles, marine vessels, and trains, as well as common materials (e.g., plastics) through off-gassing. Chemicals emitted to the air may be transported over long distances, generally in the direction of the area's prevailing winds. They can be deposited from the atmosphere to land and water surfaces through wet deposition (precipitation) or dry deposition (as particles).

Air pollutants can enter water bodies through either direct or indirect deposition. Direct deposition occurs when particulates with adsorbed chemicals are deposited onto the surface of a water body and then settle to the bottom, becoming part of the sediment. Indirect deposition occurs when chemicals are first deposited on land or other water bodies in the watershed and then transported to the water body via surface water or stormwater runoff, either in dissolved form or adsorbed to solids in the runoff, which is discharged to the water body and ultimately transported to bottom sediments. Many air pollutants deposited through direct or indirect atmospheric deposition in aquatic systems, such as the LDW, have the potential to contaminate sediment because they are hydrophobic and tend to adhere to sediment particles (PSCAA 2003). The potential sources of chemicals to the LDW through atmospheric deposition are discussed in further detail in Sections 9.3.2.4 and 9.4.2.

9.1.2 Surface water inputs

Potentially contaminated surface waters and suspended sediment may be transported to the LDW from the upper reaches of the Green/Duwamish River, tributary creeks, and through open-channel drainage ditches. The inputs from these sources vary greatly during the year; the upper Green/Duwamish system is variable because it can be influenced by a large area of mixed industrial, commercial, residential, and agricultural lands. Accidental spills and leaks (e.g., hazardous material releases from fuel storage tanks, automobiles, or other equipment) that occur upriver or along its tributaries also have the potential to enter the LDW in areas lacking collection systems, through which spills and leaks can flow directly over impervious surfaces and drain into the LDW. Surface water inputs into the LDW are discussed in further detail in Section 9.4.3.

9.1.3 Direct discharges

In general, direct discharge systems include municipal or other publicly owned drainage systems, privately owned and managed drainage systems, and sanitary/combined sewer systems. Since 2003, over 250 individual discharge points (e.g., pipes) into the LDW study area have been identified (Herrera 2004a). These individual points, often referred to as outfalls, are defined here as locations of discharge of storm water and/or industrial wastewaters transported via a collection system; though most industrial discharges are now routed through the sanitary sewer and no longer discharge directly to the waterway. The classifications of direct discharges include CSOs, emergency overflows (EOFs), and public and private storm drain systems. Direct discharge systems and the various types of outfalls are discussed in further detail in Section 9.4.4 and Appendix H.

9.1.4 Spills and leaks

Chemicals released through spills and leaks to soil, other ground surfaces (such as roadways), surface water, or groundwater have the potential to migrate to the LDW. Leaks can occur from various sources including pipes and storage tanks, industrial or commercial equipment, and process operations. Spills can occur accidentally during vehicle fueling and maintenance, or purposefully in the case of illegal dumping. The US Coast Guard (USCG), Ecology, and the City of Seattle each have programs to record and clean up reported spills and leaks of hazardous materials. These programs are discussed further in Section 9.4.5.

9.1.5 Groundwater migration

Groundwater flow in the greater Duwamish Basin is generally towards the LDW, although the direction varies locally depending on the nature of subsurface materials, hydrostratigraphy, and proximity to the LDW (see Section 2.5). Near the LDW, tidal action can greatly alter groundwater flow directions, rates, and water quality. The determination of whether a chemical identified in groundwater will reach sediment and surface water in the LDW study area is a complex process. Groundwater issues in the LDW area are discussed further in Sections 9.4.6, and an overview of monitoring information for select facilities is presented in Appendix I.

9.1.6 Bank erosion

Shoreline banks are susceptible to erosion by vessel wakes, wind waves, surface water runoff, tidal exchange, and groundwater discharge. Shoreline armoring and vegetation reduce, but do not eliminate, bank erosion. Bank slope and bank soil properties are also major factors in the susceptibility of bank areas to erosion; steeper banks are more susceptible for a given grain size. Currently, the majority of the LDW shoreline is armored with constructed steel, wood, concrete bulkheads, sheet pile walls, and riprap revetments, which reduce the potential for bank erosion in most areas. Unarmored areas exist along the banks of Kellogg Island, the shoreline east of the island, and areas to the south near the Upper Turning Basin. Although the unarmored areas are more susceptible to erosion, armored areas are also susceptible, under some circumstances. In addition, shoreline armoring is generally not designed to be completely impervious to water; groundwater and surface water are allowed to pass through the bank materials for structural stability. This seepage has the potential to entrain soil from the bank, although properly designed armoring systems would include appropriate geotechnical filters to minimize or prevent the transport of soil through the armoring system. Along the LDW, armoring systems that are not properly designed or maintained (and that have underlying soil contamination) could result in the transport of contaminated bank soils to the LDW. In addition, if chemicals are present in shoreline bank soils that have a low affinity for sorption to soil, these chemicals may dissolve into groundwater as it passes through the bank materials.

Erosion of contaminated bank material has been identified as a potential pathway to LDW sediments in a few areas along the LDW shoreline, as discussed in Section 9.4.7. Appendix I presents available bank soil data that have been collected from several properties along the LDW. Bank erosion could also be a potential pathway to LDW sediments in other areas; however, because bank material along much of the LDW shoreline has not been well characterized, the importance of this pathway from an LDW-wide perspective is not well known. Therefore, the extent of bank contamination is a potential data gap for site-specific source control evaluations in the LDW.

9.2 LDW SOURCE CONTROL ACTIVITIES

Ecology is the lead entity for implementing source control in the LDW. Ecology works in cooperation with local jurisdictions and EPA (together forming the SCWG, which is discussed in Section 9.2.1) and conducts several different source control activities within the LDW. Source control investigations assess the potential sources and pathways of chemicals in a specific area. Ecology's overall source control strategy for the LDW is discussed in Section 9.2.2. The purpose of the LDW source control program is to control sources of chemicals to LDW sediments in coordination with sediment remediation (Ecology 2004b). Ecology's source control investigation findings and their plans for implementing source control activities are documented in data gaps reports and SCAPs (see Section 9.2.3).

9.2.1 Source Control Work Group activities

Ecology works with several other entities to create and implement source control strategies and prioritize cleanup efforts in the LDW. In 2002, these entities formed the LDW SCWG. As discussed in Section 9, the members of the SCWG are public agencies responsible for source control in the LDW, including Ecology, the City of Seattle, the City of Tukwila, King County, the Port of Seattle, and EPA. The purpose of the SCWG is to share information, discuss strategies, develop action plans, jointly implement source control measures, and share progress reports on source control activities in the LDW area. Other public entities with related source control responsibilities include PSCAA and Seattle-King County Department of Public Health. Both of these entities actively coordinate with the SCWG.

9.2.2 Ecology's source control strategy

Ecology is the lead agency responsible for source control in the LDW. In January 2004, Ecology issued the final *Lower Duwamish Waterway Source Control Strategy* (Ecology 2004b). This strategy is consistent with sediment source control protocols described in EPA guidance (2002c) and the SMS (Ecology 1995). The strategy describes the process and timing for implementing source control, the roles of various regulatory agencies responsible for conducting source control (e.g., SCWG) and those providing enforcement, and Ecology's methods for tracking and documenting source control progress in the LDW.

The focus of Ecology's source control strategy for LDW sediment is to meet sediment cleanup goals and to prevent post-remediation recontamination to levels that exceed the SMS or other LDW sediment cleanup goals that will be established in the ROD (Ecology 2004b). The strategy was designed to identify and manage sources of chemicals that could contaminate LDW sediments, and it serves as the basis for the development of a series of detailed SCAPs for priority action areas.[141] Although not explicitly addressed in the SMS, Ecology is also evaluating VOCs in porewater as part of source control efforts because if VOCs are present in sufficiently high concentrations, they can cause adverse effects to benthic organisms.

The source control strategy established the priority for source control activities in the LDW according to the following four tiers (Ecology 2004b):[142]

- **Tier One –** Source control work associated with the identified EAAs
- **Tier Two –** Source control work associated with sediment cleanup areas identified for final or long-term cleanup through the RI process or in the EPA ROD
- **Tier Three –** Source control work associated with drainage basins discharging to LDW sediments that have not been identified for Tier One or Tier Two source control activities through the RI/FS process
- **Tier Four –** Source control work associated with sediment areas that have been remediated and were subsequently recontaminated above SMS criteria or LDW cleanup goals based on post-cleanup monitoring

In 2003, seven sediment areas were identified as candidate sites to be given priority for investigation and potential cleanup in the LDW because of the relatively high chemical concentrations in sediment (Windward 2003e) (Table 9-1, Map 9-2). These seven areas (candidate EAAs) were subsequently adopted as the areas included in the Tier One classification. These sediment areas were identified based on both risk- and management-based criteria and are discussed in a LDWG technical memorandum (Windward 2003e) that identified the candidate sites for early action. Risk-based criteria were based on CSL exceedances or the top 95th percentile of PCBs based on the

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[141] Ecology and EPA may revise the source control strategy in the future. Newer versions would supersede previous versions of the strategy.

[142] As of February 2008, Ecology identified 23 source control areas for evaluation in the LDW. These areas were defined in part based on drainage to the LDW. At this point, source control areas encompass almost all of the properties immediately adjacent to the LDW study area and their associated sub-drainage basins.

results of the Phase 1 risk assessments. Management-based criteria included: 1) consistency with methods used by EPA and Ecology to identify contaminated sites requiring remediation, 2) ability to prevent unacceptable recontamination at the site, and 3) consistency between potential early action cleanup methods and remedial alternatives that are likely to be evaluated for the LDW (Windward 2003e).

Table 9-1. Candidate EAAs identified in 2003

EARLY ACTION AREA NO.	ASSOCIATED NAME	RIVER MILE	SIDE OF LDW
EAA 1	Duwamish/Diagonal	0.4 to 0.6	east
EAA 2	Early Action Area 2 (RM 2.2)	2.2	west
EAA 3	Slip 4	2.8 to 2.9	east
EAA 4	Boeing Plant 2/ Jorgensen Forge	2.9 to 3.7	east
EAA 5	Terminal 117	3.5 to 3.7	west
EAA 6	Early Action Area 6 (RM 3.8)	3.8	east
EAA 7	Norfolk	4.8 to 5.0	east

Source: Windward (2003e)
EAA – early action area
RM – river mile

Some of the EAA sediment boundaries shown on Map 9-2 and subsequent maps are preliminary and have not been finalized by EPA and Ecology. The Duwamish/Diagonal and Norfolk boundaries represent the areas where removal and capping actions have been completed by King County (in both EAAs) (King County 1999g; King County et al. 2005a) and by Boeing (in the Norfolk EAA only). The T-117 boundary was identified in the draft T-117 engineering evaluation/cost analysis (EE/CA) as the removal area; the final boundary is contingent on supplemental sampling (Windward et al. 2008). The Slip 4 boundary represents the final sediment removal area delineated in the Slip 4 EE/CA (Integral 2006) and finalized in EPA's action memo (EPA 2006a). The in-water boundary adjacent to Boeing Plant 2 represents the final sediment removal area identified under RCRA (Geomatrix and Floyd|Snider 2008) for Boeing's portion of the Boeing Plant 2/Jorgensen Forge EAA. The in-water boundary adjacent to Jorgensen Forge represents the proposed sediment removal area identified under CERCLA (EPA 2008b) for the Jorgensen Forge portion of the Boeing Plant 2/Jorgensen Forge EAA. The EAA 2 and RM 3.8 boundaries are preliminary and have not been finalized by EPA or Ecology. Detailed information on these EAAs is included in Appendix I.

The seven candidate EAAs proposed by LDWG were the basis for the designation of the Tier One source control areas (SCAs) established by Ecology; however, Ecology's SCAs encompass a larger area adjacent to and upland of the LDW (Map 9-3). The SCAs associated with the EAA boundaries include the drainage basins that drain to these areas.

In 2007, Ecology identified eight additional SCAs for priority evaluation (Map 9-2) (Ecology 2007j). Identification of the eight additional SCAs was based on surface sediment data and general knowledge about the upland areas draining to those particular areas, including drainage infrastructure and facilities operating within them (Ecology 2007j). The sediment areas associated with SCAs shown on Map 9-2 correspond only to upland source control drainage areas, and do not correspond to areas that have been identified for in-water remediation. Sediment management areas will be mapped and evaluated in the FS for cleanup with final determinations by EPA and Ecology documented in the ROD.

In February 2008, Ecology expanded the boundaries of many of the 15 previously identified areas and added 8 additional SCAs (Map 9-3) (Ecology and SAIC 2008). The updated boundaries established for the 23 SCAs were developed to ensure that source control would be conducted for the entire study site. These areas were delineated based on storm drainage sub-basins and upland property boundaries. The two northernmost SCA boundaries extend beyond the northern boundary of the LDW study area because of upland drainage features relevant to source control in those areas. For many of the SCAs, a determination of whether remediation will be required for the sediments adjacent to the boundary of the SCA has not been made.

Upland parcels and drainage areas in association with these 23 SCAs have been identified by Ecology; the upland parcels associated with these SCAs are identified on Map 9-3 and in Appendix I. The 23 SCAs currently established by Ecology within the LDW are listed in Table 9-2 along with the status of the SCAP for each area. As source control investigations within these areas continue, Ecology may revise the SCA boundaries (Good 2008). Where an individual SCA falls under Ecology's tiered classification (as established by Ecology's source control strategy) will not be determined until the RI/FS is complete and the necessary level of remediation has been established (Ecology and SAIC 2008).

Table 9-2. Ecology's 23 SCAs and the SCAP status

PROPERTY NAME OR DESCRIPTION[a]	APPROXIMATE LOCATION	STATUS OF SCAP	FINAL SCAP PUBLICATION DATE[b]
Spokane Street to Kellogg Island	RM 0 to RM 1.0 west	TBD	TBD
Kellogg Island to Lafarge Cement[c]	RM 1.0 to RM 1.3 west	TBD	TBD
Glacier Bay	RM 1.3 to RM 1.6 west	completed	November 2007
Terminal 115[c]	RM 1.6 to RM 2.1 west	TBD	TBD
Trotsky Inlet (EAA 2)[c]	RM 2.1 to 2.2 west	completed	June 2007
Riverside Drive	RM 2.2 to RM 3.4 west	TBD	TBD
Terminal 117 (EAA 5)	RM 3.4 to RM 3.8 west	completed	July 2005
Sea King Industrial Park[c]	RM 3.8 to RM 4.2 west	TBD	TBD
Restoration areas	RM 4.2 to RM 4.8 west	TBD	TBD
Spokane Street to Ash Grove Cement[c]	RM 0 to RM 0.1 east	completed	June 2009

PROPERTY NAME OR DESCRIPTION[a]	APPROXIMATE LOCATION	STATUS OF SCAP	FINAL SCAP PUBLICATION DATE[b]
Duwamish/Diagonal Way (EAA 1)	RM 0.1 to RM 0.9 east	completed	December 2004
Slip 1	RM 0.9 to RM 1.0 east	completed	May 2009
King County lease parcels[c]	RM 1.0 to RM 1.2 east	TBD	TBD
St. Gobain to Glacier Northwest[c]	RM 1.2 to RM 1.7 east	completed	June 2009
Slip 2 to Slip 3	RM 1.7 to RM 2.0 east	completed	June 2009
Slip 3 to Seattle Boiler Works[c]	RM 2.0 to RM 2.3 east	completed	April 2009
Seattle Boiler Works to Slip 4[c]	RM 2.3 to RM 2.8 east	completed	June 2009
Slip 4 (EAA 3)	RM 2.8 east	completed	July 2006
Boeing Plant 2/Jorgensen Forge[c] (EAA 4)	RM 2.8 to RM 3.7 east	completed	November 2007
Boeing Isaacson/Central KCIA (EAA 6)[c]	RM 3.7 to RM 3.9 east	completed	March 2009
Slip 6	RM 3.9 to RM 4.3 east	completed	September 2008
Boeing Developmental Center[c]	RM 4.3 to RM 4.9 east	TBD	TBD
Norfolk CSO/SD (EAA 7)[d]	RM 4.9 east	completed	September 2007

Sources: Ecology and SAIC (2008), (Good 2008), (Good 2009)

[a] List of SCAs and associated names are from Ecology; some of the EAA names have been altered since the EAAs were originally selected in 2003.

[b] SCAP schedule taken from the LDW source controls status report for July 2007 to March 2008 and updated based on personal communications from Ecology (Ecology and SAIC 2008) and modified with updated information received from Ecology (Good 2008).

[c] Company names are used only to designate areas and are not necessarily included in the cleanup area or necessarily responsible for contamination in these areas.

[d] A City pump station EOF also discharges through this outfall.

CSO – combined sewer overflow
EAA – early action area
KCIA – King County International Airport
RM – river mile
SCA – source control area
SCAP – source control action plan
SD – storm drain
TBD – to be determined

Ecology is in the process of evaluating each SCA for potential pathways and sources of contamination to the LDW. Ecology assesses the status of and need for source control within each SCA by reviewing existing information, identifying data gaps, and then developing a SCAP for each area.

9.2.3 Data gaps reports and source control action plans

Data gaps reports are prepared by Ecology to summarize available information that may have implications for source control for each of the SCAs.[143] Elements of these reports include:

- Location and physical characteristics of the SCAs

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[143] Instead of data gaps reports, a series of property assessments were prepared for EAA-1, EAA-3, and EAA-5.

- Documentation of readily available information with implications for sediment recontamination
- Identification of potential COCs for source control evaluation and assessment of potential pathways
- Identification of critical data gaps that need to be addressed prior to sediment cleanup

Once the existing information and data gaps are identified for an SCA, Ecology then develops a SCAP for that area. Ecology is producing SCAPs for all 23 SCAs. The SCAP for each of these areas identifies potential chemical sources and actions needed to control them and evaluates whether ongoing sources could recontaminate sediments after cleanup. In addition, the SCAPs describe source control actions that are planned or currently underway and sampling and monitoring activities that will be conducted to identify additional sources.

Ecology has completed or is in the process of developing detailed SCAPs and data gaps reports for the SCAs.[144] As they are completed, the data gaps and SCAPs are provided to EPA, the SCWG, and stakeholders. After the source control actions identified in a SCAP have been implemented, the success of the plan will be evaluated by Ecology through a series of source control effectiveness and completeness determinations (Ecology 2004b). These evaluations will then be provided to EPA for comment and concurrence, at which time a final determination of completeness will be issued.

9.3 HISTORICAL SOURCES AND PRACTICES IN THE LDW

In the course of over 100 years of growth and development, industrial practices and waste streams have contributed chemicals to the LDW. Understanding these various historical practices is important to identify, characterize, and control sources of sediment contamination. The following section provides an overview of the industrial development of the waterway, including the dynamic reshaping of the river and the manufacturing and commercial operations that were established along its shores. The section also discusses some of the manufacturing and waste handling and disposal practices prevalent in the earlier history of the waterway. This type of information is important to consider in source identification and control effects.

Maps 9-4a through 9-4d provide information on some of the historical operations identified in the LDW study area. The majority of source information used to prepare the maps was compiled from Foster (1945) and the Pollution Control Commission (1955). Some historical information, at a more limited level, was obtained from other

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[144] At the time of this RI, a separate data gaps report had not been prepared for the Duwamish/Diagonal Way SCA; the information typically provided in data gaps reports is included in the SCAP for this SCA.

source documents, including the data gaps reports and SCAPs. The locations of the historical operations presented on the maps are the approximate locations based on available information; in many cases, the exact locations were unknown. These maps are intended to give general locations of historical operations relative to the LDW and should not be used to associate these operations with individual parcels.

Information in this section was based on readily available resources, most notably Foster (1945) and Sweet Edwards (1985). Other sources include Ecology's data gaps reports and SCAPs, other summaries of source information, and an Environmental Data Resources, Inc. (EDR), document that was prepared in 2006 for the LDW (EDR 2006b). None of the resource information has been independently verified beyond the citations noted, and LDWG cannot verify the completeness or accuracy of source documentation. Furthermore, the primary data have had varying levels of QC and data validation, which have not generally been documented in the literature reviewed. Therefore, information included in this section should not be relied upon as complete. Ecology, the lead agency for source control, may require additional site-specific information or determine that a more thorough evaluation of the data is needed. It is expected that more thorough evaluations of the data will be conducted in the course of further source control investigations.

9.3.1 Industrial development of the LDW

Historically, land adjacent to the Duwamish was accessed using the river as a transportation route. By 1874, Seattle’s first railroad had been established, allowing further access to and development of lands along the river. Between 1901 and 1913, the LDW, the East and West Waterways, and Harbor Island were all developed through dredging, filling, and channelization of the Duwamish River and its delta (Section 2.2). Filling of mudflats and wetlands along the LDW created additional land that was available for development, and dredging and channelization of the LDW allowed oceangoing ships and barges access to the LDW. A steam plant used for electric power generation was built along the LDW in 1906.

Shipyards, airplane manufacturing, cement manufacturing, food processing, and cold storage were early industries along the LDW that became increasingly important with the onset of World War I (Sato 1997). Other common industries at this time included lumber storage and milling yards, metal fabrication, and equipment manufacturing (Sato 1997; Ecology 2006a). Residential areas in Georgetown and South Park began to grow with the increase in job opportunities and improved transportation; the South Park neighborhood area increased by more than 66 ac after channelization of the river, and the new land was claimed by farmers (Wilma 2001b).

The aviation industry continued to develop when Boeing Field (now KCIA), Seattle’s first municipal airport, was opened in 1928 (Wilma 2001a). Thousands of B-17 and B-29 bombers were produced at Boeing Plant 2 during World War II (King County 2005c). Boeing has had several other facilities near the LDW at various times. World War II also resulted in an increased need for shipyards and maintenance docks for the

US Navy. Harbor Island became home to several ship-building, salvaging, and maintenance companies, and it became an important port for economic trade. A lead smelter was also built on the island in 1937 (Tetra Tech 1988b).

Construction and cement companies, slaughterhouses, meat packing facilities, and a cannery were also located along the LDW in the 1930s and 1940s (Foster 1945). By 1945, industrial facilities were prevalent along the LDW. Lumber production continued to develop, including the use of pole treating and wood preserving processes (Foster 1945). Metal working facilities, machine shops, shipyards, cement and asphalt companies, manufacturing facilities, and meat packing companies were all common industries during the World War II era and following years. Lime and concrete plants and chemical manufacturing (including paint, glue, resin, and wood preservatives) were also established in the LDW at this time. Industrial development resulted in an increased need for waste disposal, and many casual dumping sites were created. Eventually, some dumping sites became official landfills that received industrial waste, though most remained small informal landfills used by neighboring properties for their waste disposal.

Continuing through the 1950s and 1960s, metal working facilities along the LDW included metal manufacturers and recyclers, galvanization plants, and foundries. Steel, tin, aluminum, titanium, nickel, and lead (at the Harbor Island smelter) were all used in various metal-working industries. With an increase in demand for fuel products along the LDW, several petroleum storage facilities and storage tank farms were established on Harbor Island. From the 1950s to the 1970s, several smaller companies and landfill sites along the LDW were closed and redeveloped as parking lots or other commercial and industrial facilities.

By the 1980s, much of the land adjacent to the LDW was occupied by industry. Aviation industries and Port of Seattle operations, including cargo storage and transport, continued to expand during this time. Harbor Island was still used largely for shipyard activities and petroleum storage. Other metal-working industries remained on Harbor Island and along the LDW during this time period (Sweet Edwards 1985). Wood preserving, chemical processing (including used oil and antifreeze), and other types of manufacturing continued throughout the 1980s. Industry to the south of Harbor Island near Kellogg Island was primarily related to cement and gypsum manufacturing and included an upland waste disposal site (Sweet Edwards 1985). In 1998, the Harbor Island lead smelter was permanently closed.

Many of the industries that were present along the LDW in the past are still in operation today. Currently aviation facilities used for the construction, maintenance, and operation of aircraft, Port-owned and privately owned terminals used for cargo shipping and receiving, motor vehicle and marine vessel maintenance, cement production, manufacturing plants, and various other industries all operate along the

LDW. Maps 9-4a through 9-4d show approximate locations of selected historical operations along the LDW.

9.3.2 Historical discharge sources

Historically, chemical disposal practices were not regulated as they are today. In many cases, chemicals and other wastes were discharged directly to the LDW or disposed of on upland properties without sufficient containment (Foster 1945; Seattle-King County Department of Public Health 1984). This section provides an overview of information regarding historical sources to the LDW.

9.3.2.1 Historical industrial discharges

Potential sources of contamination along the LDW were documented in several historical reports (Foster 1945; Sweet Edwards 1985; Tetra Tech 1988b). In the report by Foster (1945), the river was divided into three sections for assessment: the Duwamish Waterway, which received predominantly industrial waste, the Green River below the City of Palmer (south-central King County), which received primarily domestic sewage and agricultural waste, and the upper Green River, which received little contaminated discharge of any type at that time.

Wastes generated during the development and ongoing industrial history of the LDW had the potential to be discharged into the waterway. Wastes were transported to the LDW directly through storm drainage systems, as a component of the discharge from sanitary sewer systems, through atmospheric deposition, or direct disposal along the shoreline. Sanitary sewer systems serviced large industrial areas along the LDW. Information on sewer system discharges are discussed in Section 9.3.2.2.

Historically, many industrial facilities used unlined, open-air waste ponds for pre-treatment of slurries and other waste water (Sweet Edwards 1985; Foster 1945). Often the ponds and lagoons were located along the LDW shoreline and contained chemicals that could enter area soil and/or groundwater and eventually enter the LDW. Waste slurries derived from routine maintenance and cleaning of tanks and equipment were often dumped directly into the LDW or into wastewater ponds on a daily to yearly basis, depending on the industry (Foster 1945).

Several lumber yards were located along the LDW in the past, particularly at the upstream reach of the LDW near the Upper Turning Basin. Prior to 1945, considerable amounts of sawdust were disposed of in the LDW (Foster 1945); after that time, sawdust became valued as a fuel product and was kept by most mills. Waste products from lumber mills and sand and gravel companies were sources of solids (in addition to ongoing inputs from the Upper Duwamish/Green River) to the LDW in the past. Additional wastes associated with lumber processing included creosote used in pole dipping, and arsenic and sulfate salts of copper and zinc used to preserve logs. Creosote is known to contain PAHs. Other wood preservatives and resins may have contained TBT (EPA 2003).

Historically, oil and fuel were discharged to the LDW from various industrial activities including ship-building and repair facilities, ship salvaging companies, brick manufacturing, machine shops, pole treating facilities, and hydraulic equipment manufacturing (Foster 1945). Oily wastes were observed seeping from facilities located along the LDW shoreline in the past (Sweet Edwards 1985). The storage, transport, and use of fuel at aviation, railway, motor vehicle, and fuel storage/tank farm facilities were also historical sources of petroleum product wastes that may have reached the LDW through both direct discharge (spills and dumping), transport through storm drain networks, and indirect discharge (migration through soil or groundwater). Historically, recycled or waste oils sometimes contained PCBs from electrical or hydraulic equipment and other sources directly containing PCBs or due to mixing or cross-contamination with PCB oil from those sources (ATSDR 2000). Recycling of these oils often resulted in oil spills (Sweet Edwards 1985).

In the late 1970s, the EPA issued regulations banning the manufacture of PCBs. The use of PCB-containing equipment began to be phased out at this time. The presence of PCBs in waste oils might have diminished after these regulations were introduced in the late 1970s. However, the timing and amount of the reduction of PCBs in waste oils are uncertain.

Significant uses of PCBs were as a dielectric fluid in transformers and as a component of hydraulic oils; releases to the environmental associated with both of these uses have been well documented. PCBs have also been used in many other products, such as capacitors, heat transfer systems, inks and carbonless copy paper, the ballasts of some fluorescent light fixtures, and paints. PCB contamination along the LDW has also been documented in connection with the waste oil used as fuel in asphalt manufacturing (Ecology 2005a; Windward et al. 2005b), as well as associated with caulking and sealant materials used on pavement and asphalt (SAIC 2007e).

Industrial waste discharges originating from manufacturing and metalworking companies often contained various chemicals. Acidic solutions were among the most widely reported chemical releases associated with these industries (Sweet Edwards 1985; Foster 1945). Other wastes included metal scraps, sodium borate, cyanide, and zinc salts (Foster 1945). Chromium, copper, cadmium, and nickel are associated with electroplating (Sweet Edwards 1985). In addition, the Harbor Island and Asarco smelters emitted lead, arsenic, and other metals to the atmosphere (Huey 2002; Ecology 2007b; Area-Wide Soil Contamination Task Force 2003) (Section 9.3.2.4).

Shipbuilding and marine vessel maintenance is another industry that has been present in the LDW for many years. Metals (including arsenic, chromium, copper, lead, and zinc) and petroleum products were common waste products produced by shipyards activities (Tetra Tech 1988b). Anti-fouling paint applied to boats often contained TBT, copper, mercury, and other metals (SAIC 2007d). Sandblasting, painting, boat repair, and other maintenance activities also had the potential to contribute metals and various petroleum products to the LDW. In addition, solvents, acids, various types of paint, antifreeze, and other chemicals have also been associated with shipyards (SAIC 2007d).

Several waste products are associated with the aviation industry, including alkaline carbide wastes from acetylene generators (Foster 1945), petroleum products used for fuel, and metal wastes produced in the manufacture of aircraft (Foster 1945; Ecology 2006a). Solvents and paints were also associated with aircraft maintenance.

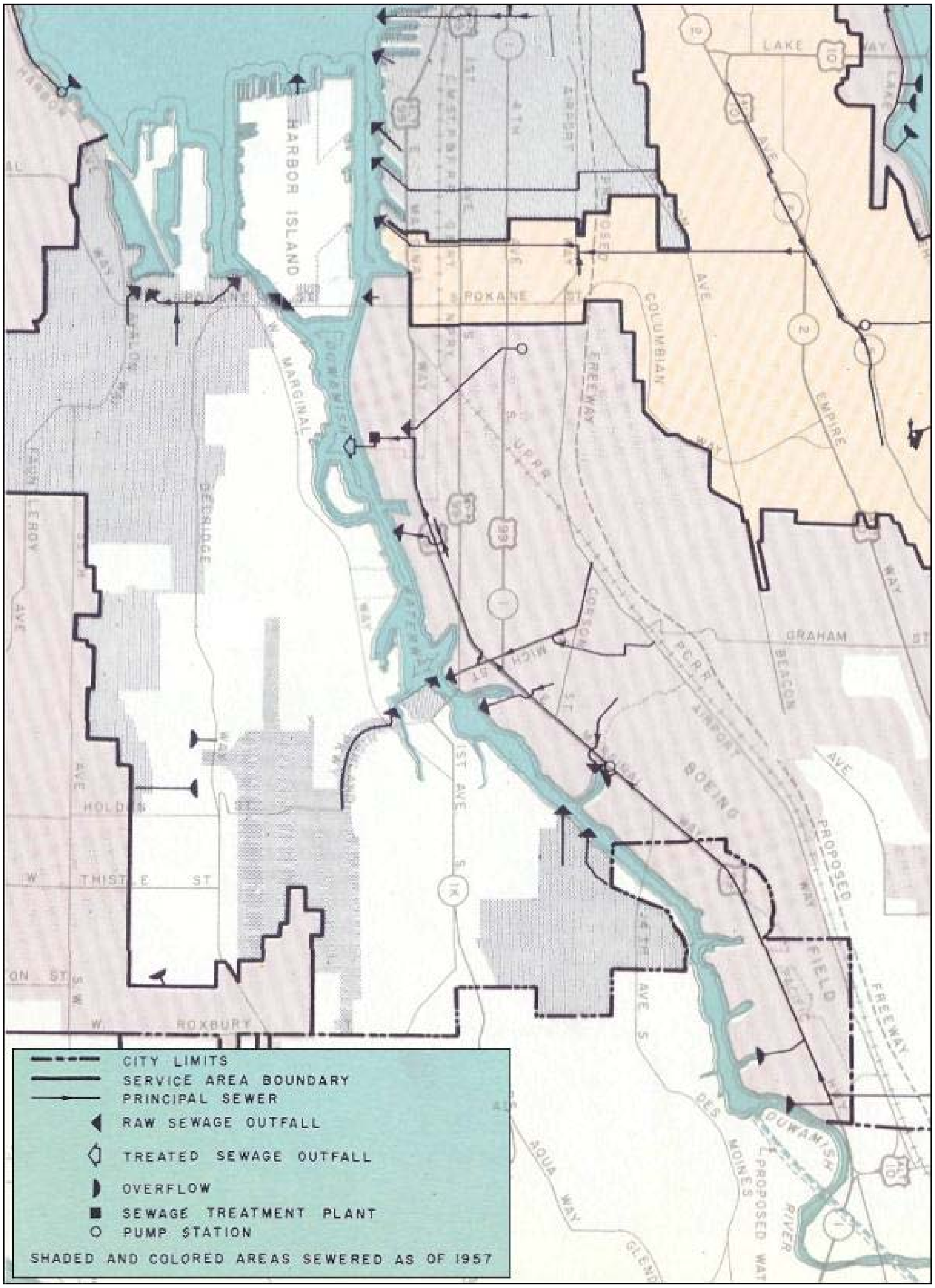
Chemical manufacturing has produced various wastes that entered the LDW through a number of pathways in the past (Sweet Edwards 1985). Used antifreeze is generally known to contain lead and cadmium, and production of adhesives, water-soluble glues, formaldehyde, and wood-preserving resins produced phenol formaldehyde and pentachlorophenol waste (SAIC 2007d). Dioxins and furans are known to be a by-product of pentachlorophenol production and fires and incineration of both chlorinated and non-chlorinated organic compounds. Copper was a waste product associated with the production of vanillin and wood preserving resins (Sweet Edwards 1985). Cement and asphalt manufacturing produced several waste products, such as cement kiln dust, which were often disposed of at casual landfill sites (Sweet Edwards 1985).

Organic wastes originating from various operations were also commonly discharged directly to the LDW in the past. Agricultural operations, slaughterhouses, meat packing facilities, and raw sewage discharges were all sources of organic wastes, including blood, manure, and grease (Foster 1945). Historical sewage discharges are discussed further in Section 9.3.2.2.

9.3.2.2 Historical sewer system discharges

Prior to the creation of Metro (now the King County Wastewater Treatment Division) in 1958, local sewer systems along the waterway discharged raw sewage directly into the LDW. These raw sewage discharges potentially contained mixed industrial wastes originating from facilities along the waterway. Although the LDW is currently served by a combination of separated, partially separated, and combined sewer systems, historically, most of the LDW area was served by a combined storm/sanitary sewer system. Some small areas in unincorporated King County along the river remain on septic systems. The extent to which industry along the river uses septic systems is unknown.

As shown on Figure 9-2, approximately seven of these combined sewer/sanitary outfalls (raw sewage outfalls) and five CSOs (overflows) discharged to the LDW in 1958. The combined sewer/sanitary outfalls discharged raw sewage, stormwater, and industrial waste streams directly to the waterway in areas not served by a wastewater treatment plant. CSOs discharged similar wastewater but only when the capacities of the treatment plants or their conveyance systems were exceeded.



Source: Brown and Caldwell (1958)

Figure 9-2. Outfalls that historically discharged raw sewage to the LDW

The Henderson-E Marginal Way S system collected stormwater runoff and municipal/industrial wastewater from approximately 5,900 ac (on the east side of the waterway and conveyed flow to the Diagonal Avenue S wastewater treatment plant (WWTP). The Diagonal Avenue S WWTP operated from 1938 until approximately 1969 when the plant was closed and flows were diverted to the West Point WTTP. In

1958, the total population served by the Diagonal Avenue S WWTP was about 30,000 (Brown and Caldwell 1958). Two of the combined raw sewer outfalls and all five of the CSOs in the LDW identified on Figure 9-2 were associated with the Henderson-E Marginal Way S system. The last sanitary sewer outfall of this system was connected to the West Point system in 1976.

In 1958, the West Seattle system served an area of about 1,950 ac located along the west side of the LDW (Brown and Caldwell 1958). At that time, none of the raw sewage, stormwater, or potential industrial waste streams from this system was treated. Four raw sewage outfalls discharged to the LDW study area and another three outfalls discharged outside the LDW in the vicinity of West Waterway (see Figure 9-2). In addition to the areas served by the West Seattle system, large areas on the west side of the river, stretching from the southern tip of Harbor Island to approximately RM 2.8, were not connected to the City of Seattle's sanitary sewer system as of 1958 (Brown and Caldwell 1958). These areas relied on onsite systems (e.g., septic tanks and drain fields) to infiltrate wastewater into the underlying soil in order to dispose of sanitary and industrial waste. However, some private systems used dry wells or directly discharged to roadside ditches or into surface water bodies (Brown and Caldwell 1958).

As mentioned previously, the first WWTP along the LDW, the Diagonal Avenue S WWTP, was built by the City of Seattle in 1938 near RM 0.6, approximately 1,200 ft south of the current location of the Diagonal Avenue S CSO/SD (Brown and Caldwell 1958). The Diagonal Avenue S WWTP provided primary treatment for sanitary wastewater, industrial waste streams, and stormwater collected from the Rainier Beach neighborhood near Lake Washington and the industrial district along E Marginal Way S from the Diagonal Avenue S WWTP south to the Upper Turning Basin (the area previously identified as the Henderson-E Marginal Way S system). Industrial facilities along the LDW were not required to redirect industrial waste discharges to this system until the early 1950s.

The Diagonal Avenue S WWTP had a design capacity of 8 million gal. per day (mgd). Regulating stations were installed in the collection system (e.g., at the present location of the Norfolk CSO/SD and the present location of the E Marginal Way S pump station in Slip 4) to keep the WWTP from overloading. In 1956, the dry weather flow to the Diagonal Avenue S WWTP ranged from 2.5 to 5.0 mgd. However, up to 90% of the wet weather flow was bypassed directly to the LDW because of limited capacity at the WWTP and the fact that the interceptor sewers were too small to convey the required flow from the trunk lines (Brown and Caldwell 1958).

The principal source of organic waste to the LDW was raw sewage that originated from industrial facilities and residential areas along the LDW (Foster 1945). This type of waste was usually routed through the City's sewer system into the Diagonal Avenue S WWTP for example; however, many of the industrial properties along the LDW were too low in elevation for effective use of the gravity fed sewer lines and raw sewage and industrial waste streams in these locations were either stored temporarily in open ponds or discharged untreated to the LDW (Brown and Caldwell 1958).

By the late 1950s, industries that discharged wastewater (industrial or sanitary waste) to the public sewer system or directly to a nearby receiving water body were required to obtain a wastewater discharge permit from the Pollution Control Commission. At that time, 150 industries in the City of Seattle had industrial discharge permits. A sewerage and drainage survey conducted in 1957 indicated that industrial waste streams that discharged directly into the LDW without pre-treatment could potentially reach a volume of about 33,900 gal. per day (gpd). The majority of this industrial waste (63%) was associated with metal-related industries (Brown and Caldwell 1958). The study indicated that the wastes from metal-plating industries were particularly hazardous and required separate pretreatment.

In the same survey, a total of 14 industries that discharged industrial wastewater (at a rate of approximately 2,500 gpd) into networks that would be treated at the Diagonal Avenue S WWTP were also identified (Brown and Caldwell 1958). Industrial discharges that passed through the Diagonal Avenue S WWTP originated from adhesives and chemical plants, metal plating facilities, beverage bottling operations, cement handling and production, compressed gas operations, food canning, sawmill and wood-working related industries, steel plants, and truck manufacturing operations (Brown and Caldwell 1958).

In 1962, Metro assumed operation of the Diagonal Avenue S WWTP until its closure in 1969, when the Elliott Bay Interceptor (EBI) was brought on-line and flow was diverted to the West Point WWTP, which discharges treated wastewater to Puget Sound. The evolution of the wastewater permit programs required industrial properties along the waterway to upgrade their systems and interconnect with the interceptor line.

Starting in 1969, large portions of the public and private sewer lines were connected into the EBI line, and subsequent system overflows were eventually limited to the current King County and City of Seattle CSO locations (see Section 9.4.4.4). Some connections required complex infrastructure improvements; thus, raw sewage was discharged for several years after 1969, including through the Henderson/E Marginal Way S system. The raw sewage outfall into Slip 4 continued to discharge until 1976 (City of Seattle 1998).

9.3.2.3 Significant known historical spills

In 1974, a spill occurred at the GSA dock when a transformer was dropped and cracked while being loaded onto a barge in Slip 1, resulting in the release of approximately 980 L (260 gal.) of high-concentration PCB dielectric fluid (primarily Aroclor 1242) into the LDW (EPA 1975a). Two separate dredging operations were conducted to address the sediment contamination that resulted from the spill. An initial cleanup by EPA in 1974 was estimated to recover approximately 300 L (80 gal.) of PCBs.

A second dredging attempt by EPA and the USACE in 1976 was required to recover PCBs that had spread throughout Slip 1 and into the river channel, in part because of a 20-yr flood that occurred in the winter of 1975-1976. The second cleanup involved hydraulic dredging of sediments containing PCBs in the northwest corner of Slip 1. About 38 million L (10 million gal.) of PCB-contaminated slurry was piped to settling lagoons on the Chiyoda Property (formerly the Diagonal Avenue S WWTP property and currently the location of the Port of Seattle's T-108 (Map 9-4a). Most of the slurry was deposited in one of the lagoons located closest to the LDW, while the second lagoon received overflow water from the first lagoon. Water pumped from the lagoons was filtered through a sand and charcoal filter to remove suspended particles and PCBs before discharge to the LDW. It was estimated that the second dredging removed another 640 L (170 gal.) of the 980-L (260-gal.) spill (EPA 1975a).

Two other known significant historical spills were documented in the Washington State Pollution Control Commission (WPCC) Progress reports (WPCC 1945-1957, as cited in Dexter et al. 1985). Dexter et al. (1985) did not report the exact years of the spills, but both were reported to the WPCC between 1945 and 1957. The Duwamish Manufacturing Company reportedly spilled 5,000 gal. of crude oil from a valve left open on a tank (the exact date and location of the spill is unknown). The Duwamish Manufacturing Company was located near RM 3.5 W (Map 9-4c). The site was subsequently cleaned, but most of the leaking oil had already entered the LDW. Another reported incident occurred at the Duwamish/Diagonal Avenue S SD, which is located at approximately RM 0.7 E (Map 9-4a). A large amount (quantity not reported) of oil was found entering the LDW from the SD. The oil had come from the Union Pacific Railroad roundhouse where drained oil had overflowed from a sump, which was intended to prevent the oil from reaching the sewer (WPCC 1945-1957, as cited in Dexter et al. 1985).

9.3.2.4 Atmospheric sources

Historical atmospheric sources may have contributed to sediment contamination in the LDW in the past. Sources of these discharges included regional smelter facilities, fossil fuel combustion by motor vehicles and industrial facilities, cement manufacturing plants, and other industrial and urban practices. Air emissions were not commonly monitored or regulated prior to the establishment of the Clean Air Act (CAA) in the 1970s.

Regional Smelter Facilities

The LDW is located in an area that has been designated as having potential soil contamination caused by historical smelter emissions (Area-Wide Soil Contamination Task Force 2003). Two historical regional smelter facilities were potential sources to the LDW. The Asarco copper smelter operated from 1890 to 1985 in Ruston, Washington, northwest of Tacoma. A lead smelter was also located on Harbor Island from the 1930s until 1998, though operations were greatly reduced in the mid-1980s.

Smokestack emissions from the Asarco smelter contained arsenic and lead, and were likely a historical source of these metals to the LDW (Ecology 2002c). Prevailing winds are a large factor in determining the spread of smelter emissions (Ecology 2001b); for much of the year, prevailing winds in the LDW are from the south (PSCAA 2007c) where the Asarco smelter was located. Heavy metals were distributed through atmospheric deposition to over 1,000 square miles of the Puget Sound basin from the Asarco smelter. Map 9-5 provides information on the aerial size and extent of the arsenic plume footprint from the Asarco smelter and ranges of arsenic contamination in surface soil (Ecology 2007b).

Since the closure of the Asarco smelter, Ecology and several local health departments have initiated studies to outline the extent of the plume "footprint" by testing arsenic and lead concentrations in area soils. Soil samples collected to date show that arsenic concentrations in the LDW drainage basin range from 17 to 30 mg/kg dw (Ecology 2001b). Lead concentrations ranged from 37 to 200 mg/kg dw. MTCA Method A soil cleanup levels for unrestricted land uses for arsenic and lead are 20 and 250 mg/kg dw, respectively. Soil characterization within the plume footprint is still ongoing (Ecology 2007k).

It is estimated that the Harbor Island lead smelter deposited lead through atmospheric deposition over at least 640 ac of land (Area-Wide Soil Contamination Task Force 2003). Emissions from the smelter contributed to lead contamination of the soil on Harbor Island, leading to its listing as an NPL site in 1983 (EPA 1993b). Emissions from the smelter were also likely deposited into the LDW through direct and indirect atmospheric deposition. An air monitoring station formerly located on Harbor Island was used to collect samples to analyze for lead concentrations in the air. Prior to the smelter's closure, lead was often detected in air samples at concentrations above federal standards. Quarterly average lead concentrations over 10.5 $\mu g/m^3$ were reported by PSCAA for the 1981 monitoring year (PSCAA 2006a). By 1998, all operations at the Harbor Island smelter ceased and lead concentrations in air dropped below the Quarterly Federal Standard of 1.5 $\mu g/m^3$. Air monitoring on Harbor Island was discontinued in 1999.

It is likely that both the Asarco smelter and the Harbor Island smelter contributed metals, particularly arsenic and lead, to the LDW through direct and indirect atmospheric deposition in the past. The contribution of metals to the LDW from historical smelter emissions compared with other sources has not been determined and is difficult to assess.

Historical Urban Atmospheric Sources

In addition to smelter facilities, other historical urban atmospheric sources have potentially contributed to sediment contamination in the LDW, such as wood burning, industrial processes for heating, burning of fossil fuels including emissions from motor vehicles, power generating facilities, wood treatment, waste incineration, and cement kilns.

Air pollutants commonly associated with urban activities in the past included particulate matter (PM), metals, PAHs, PCBs, and dioxins and furans. PM and PAHs were emitted from the burning of fossil fuels such as coal and oil, used in various industrial processes and for domestic purposes such as transportation and heat. Wood burning and waste incineration were other historical practices that emitted PM, PAHs, dioxins and furans, and metals (PSCAA 2003).

In the past, large quantities of lead were emitted to the atmosphere from the use of leaded gasoline, which was used primarily from the early 1920s through the 1990s, though it was largely phased out by the mid-1970s because of health concerns (Ecology 2001b). Chromium and nickel emissions are also associated with the burning of fossil fuels (PSCAA 2006a).

Other industrial processes that were common in the LDW in the past also may have contributed air pollutants. Arsenic is often emitted to the atmosphere through wood treatment processes and distillate oil combustion, and chromium is emitted by chrome electroplaters (PSCAA 2006a). Cement kiln emissions can contain chromium, mercury, lead, and zinc, as well as dioxins and dioxin-like compounds, PCBs, and PAHs (EPA 2007f). Several of the historical atmospheric sources are still present near the LDW today, though most emissions are now registered and regulated by PSCAA and EPA.

9.3.2.5 Waste disposal, fill areas, and abandoned landfills

Source information from upland disposal or abandoned landfills sites is generally not well characterized because information on the current conditions of these sites is either largely unavailable or has not been thoroughly investigated. This section summarizes readily available information that pertains to upland waste disposal and abandoned landfills in the vicinity of the LDW.

Waste Disposal and Fill Areas

Historical waste disposal and fill areas in the LDW are discussed in Sweet Edwards (1985) (Map 9-6). The focus of this report was to identify potential sources of contamination to groundwater through the leaching of wastes from these areas. Prior to the 1970s, it was common practice for industries to store or bury waste products on their property or on nearby vacant lots. Filling of the old Duwamish River and grading of new sites along the LDW occurred from approximately 1910 until the 1960s (Sweet Edwards 1985). Potential contamination from the fill material is uncertain, because the quality and source of fill material used during this time period is undocumented and largely unknown. Fill was assumed to be primarily dredge spoils from the main channel, and fill sites were likely located near dredge sites to keep transportation costs low (Sweet Edwards 1985).

Shoreline Banks

The LDW shoreline banks were used as areas for fill and refuse disposal in the past. This practice was used to dispose of wastes and reclaim land; it is likely that the majority of the LDW shoreline was used to dispose of fill material and other wastes during construction of the waterway. Currently, bank conditions range from exposed slopes of varying grade to vertical grades with armoring with constructed steel, wood, concrete bulkheads, sheet pile walls, and riprap revetments, which reduce the potential for bank erosion in most areas. Bank soil characterization data are not available for the majority of the properties along the LDW shoreline, although data exist for some properties, such as Rhône-Poulenc, Boeing Plant 2, Jorgensen Forge, Slip 4, Industrial Container Services/Trotsky property, Douglas Management Company, T-108, and T-117. Bank soil information for several properties along the LDW is discussed in Appendix I.

With the amount of data currently available, the correlation between bank soil conditions and intertidal sediment quality is not well understood. However, it is recognized that bank soils could potentially affect sediment quality directly through erosion or indirectly through tidal exchange or groundwater interactions. Indirect effects were investigated during the seep study (see Section 9.4.6) (Windward 2004b). Bank soil, in both armored and unarmored shoreline areas, could erode and potentially affect nearshore sediment quality or groundwater quality in some areas.

Abandoned Landfills

Upland landfill disposal represents another type of waste disposal practice commonly used in the past. Prior to the establishment of municipal waste dumps, from the 1800s into the first half of the 1900s, residential and industrial solid wastes in the Seattle area were commonly dumped in low-lying and wet areas. As with the shoreline areas, waste disposal in these areas served the dual purpose of providing dumping grounds for garbage and other debris while reclaiming low-lying land, including former slips within the LDW (Map 9-6), for future development. Even after the establishment of municipal waste landfills, localized landfilling at industrial and residential properties continued along the LDW. Most of the small-scale sites associated with the historical practice of landfilling are undocumented.

Burning was a common practice for the handling of disposed wastes at private residences, industrial facilities, and at some landfills, and accidental fires sometimes occurred (Seattle-King County Department of Public Health 1984).

An unknown number of such waste sites underlie portions of the city historically comprising Elliott Bay and Duwamish tidelands. These areas were used to dispose of solid and liquid wastes and dredge spoils (Map 9-6). As a consequence, there are likely numerous smaller undocumented landfills, in addition to documented municipal landfills, that are located in the LDW drainage area.

In 1984, the Seattle-King County Department of Public Health (1984) conducted a survey of some of the larger and better-documented former landfills in the city to assess public health risks at these sites. This survey concluded that in most cases abandoned landfill sites were adequately capped and not likely to pose a direct-contact risk to people. However, the survey noted a potential for exposure to leachate that has migrated via groundwater into underground piping networks or building excavations. The survey also stated that degradation could occur at abandoned landfills through the generation of methane gas as wastes decompose.

In 1986, the department followed up with a toxicity hazard assessment of some of these landfills (Seattle-King County Department of Public Health 1986). The four landfills reviewed in the LDW drainage basin included the Judkins Park Landfill, the Sick's Stadium Landfill, the 6th Avenue S Landfill, and the South Park Landfill (Map 9-6). Surface water runoff, surface soils, and subsurface gases where assessed for the presence of chemicals that could pose a risk to public health or site workers. The study reported that no significant environmental health problems were identified at the Judkins Park and Sick's Stadium Landfills, and no further study was recommended. No conclusions or recommendations were made for the 6th Avenue S Landfill, the boundaries of which are not known, because of the limited amount of information available for that site. The assessment recommended that groundwater and stream sampling be conducted at the South Park Landfill.

King County implemented an independent remedial investigation and a quarterly groundwater monitoring program at the South Park Landfill in 1997. Long-term monitoring indicated that vinyl chloride was the only COC present in downgradient wells located approximately 2,000 ft from the LDW, but that concentrations rarely exceeded MTCA Method B surface water cleanup standards. Vinyl chloride readily degrades under aerobic conditions, and with precipitation infiltration and the tidal-mixing effect expected during transport through groundwater, it is highly unlikely that vinyl chloride from the site would reach the LDW at concentrations that could influence sediment conditions (Windward 2003c).

The South Park Landfill has been a MTCA site since 1988. An agreed order is being negotiated by the City of Seattle, King County, and South Park Property Development, a private business group that owns a portion of the former landfill, to conduct an RI/FS at the South Park Landfill (Wang 2008). The RI/FS will be conducted to delineate the nature and extent of contamination at the site; this information will be used to establish the MTCA site boundary. The RI/FS process will also determine whether additional measures beyond typical landfill closure practices are necessary at the site to confirm that the former landfill will not be a source of contamination to the LDW.

Although previous investigations of abandoned municipal landfills in the LDW drainage basin have assessed these sites for risks to public health, they have not assessed the potential for these areas to act as sources of contamination to LDW sediments. In addition, smaller, undocumented landfill areas haven't been assessed for their potential to contribute chemicals to the LDW. The potential for abandoned landfills in the LDW drainage basin to affect sediment quality in the LDW is not known, and few data are available to assess the potential impacts from abandoned landfills (Sato 2008). Environmental investigations are ongoing at the South Park Landfill; abandoned landfill sites would need to be investigated in order to determine whether or not they are potential sources of contamination to the LDW.

9.4 SOURCE IDENTIFICATION AND CONTROL EFFORTS

The previous section discussed the sources and pathways of historical contamination that have the potential to affect the quality of the environment in the LDW. Identification of current and ongoing sources is equally important to promote the effective implementation of remediation strategies while ensuring that areas being remediated are not recontaminated after cleanup is complete. This section provides a detailed discussion of the known sources of contamination that could potentially affect sediment quality in the LDW.

This section discusses the ongoing source control work being carried out by various agencies as they implement many different regulatory programs. The various regulatory and public entities currently doing source control work in the LDW include the City of Seattle, the City of Tukwila, King County, the Port of Seattle, Ecology, and EPA. The different source control methods and tools being used by these entities to identify, characterize, and control sources include permits, enforcements, regulations, pollution prevention programs, inspection and maintenance programs, environmental investigations, pathway assessments, and cleanups.

9.4.1 Available source information

Numerous documents and resources were reviewed and evaluated during completion of the RI to help identify potential sources of contamination in the LDW. These resources included various federal, state, and local environmental databases that document a range of information including suspected site contamination, investigative status, and past cleanup activities. Agency and SCWG representatives provided clarification and additional updated information on the status of sites along the LDW, as necessary. The following section presents an overview of these various resources and the results of the review.

9.4.1.1 Historical documents and available reports

A considerable number of regional and site-specific source investigations and associated control actions have been performed in the LDW study area under a wide range of RCRA, CWA, and MTCA actions. Chemical loadings, sources, pathways, and control efforts within the LDW were initially compiled and reviewed by Metro in 1983 (Harper-Owes 1983) based on a synthesis of numerous investigations conducted from 1960 to 1982. The purpose of the study was to assess the water quality and sediment problems in the Duwamish estuary and to determine how these problems affected the beneficial uses of the river. A more recent data compilation was prepared by King County as part of the 1999 WQA for the Duwamish River and Elliott Bay (King County 1999e). The focus of the WQA study was to assess the risks in the LDW and identify the effects of CSO discharges into the LDW and Elliott Bay on those risks.

As discussed in Section 9.2, Ecology has published a source control strategy for the LDW (Ecology 2004b). That document outlines the major components of the source control program and describes the roles of the regulatory agencies participating in the LDW SCWG. The goal of the strategy is to meet SQS and to prevent future exceedances of SMS criteria and sediment cleanup goals established in an LDW ROD.

9.4.1.2 Environmental databases

Information on upland facilities within the LDW drainage basin was collected from regulatory databases maintained by EPA, Ecology, King County, the City of Seattle, and the Puget Sound Clean Air Agency (PSCAA). Much of the information was from a database search prepared by EDR in 2006 for the LDW study area.

Tables 9-2 and 9-3 list the various regulatory databases identified in the EDR report (EDR 2006b); information provided in the databases identified in Table 9-2 is summarized in Appendix G. The documentation provided by EDR included information on numerous facilities located within several miles of the LDW study area. Databases were selected for inclusion in the RI based primarily on their potential relevance to LDW source control. Tables 9-3 and 9-4 further summarize the rationale for determining the inclusion or exclusion of database information included in Appendix G.

Table 9-3. Databases used to identify facility information on potential ongoing sources to the LDW as detailed in Appendix G

Location in Appendix G	Database	Description	Rationale for Inclusion in Report	Source
Tables G-1 through G-3	National Pollutant Discharge Elimination System Permits	Permits are issued for industrial, municipal, and large construction area discharges to surface water bodies by Ecology under the NPDES program.	Facilities discharge directly to the LDW or upriver of the LDW.	Ecology's Water Quality Permit Life Cycle database (Ecology 2007g, f; EDR 2006b)
Table G-4	King County Industrial Waste Program Permits	King County issues permits or discharge authorizations for industries discharging into its wastewater treatment system.	Facilities discharge directly to the LDW through King County and SPU CSOs.	King County (Tiffany 2006a, b)
Table G-5	Large-Quantity Generators	EPA database that tracks facilities that generate over 1,000 kg of hazardous waste or over 1 kg of acutely hazardous waste per month.	Given the volume of potential hazardous waste generated, LQGs have the potential to be a source of contamination to the LDW.	EDR (2006b) from Ecology database
Table G-6	Confirmed and Suspected Contaminated Sites List	This list is a database used by Ecology's Toxic Cleanup Program to track progress on all confirmed and suspected contaminated sites in Washington State.	Sites with toxic chemicals present or suspected to be present in various environmental media in the vicinity of the LDW.	EDR (2006b) from Ecology database
Table G-7	Leaking Underground Storage Tank; cleanup in progress	Inventory of active LUST incident reports maintained by Ecology.	LUST sites with soil and/or groundwater contamination in the vicinity of the LDW.	EDR(2006b) from Ecology database
Table G-8	Leaking Underground Storage Tank; remediation reported as complete	Inventory of inactive (remediated) LUST reports maintained by Ecology.	LUST sites may have provided a source of contamination prior to site cleanup.	EDR (2006b) from Ecology database
Table G-9	Solid Waste Facility/Landfill	Inventory of solid waste disposal facilities or landfills, both active and inactive, or open dumps that failed to meet RCRA Subtitle D Section 4004 criteria for solid waste landfills or disposal sites.	Releases of toxic waste from disposed materials at SWF/LF facilities in the vicinity of the LDW.	EDR (2006b) from Ecology SWF/LF database
Table G-10	Lists of active and inactive drycleaners	Ecology-maintained list of registered (i.e., active) and inactive drycleaners.	Drycleaners can generate hazardous waste that can contaminate groundwater and soil in the vicinity of the LDW.	EDR (2006b) from Ecology databases

LOCATION IN APPENDIX G	DATABASE	DESCRIPTION	RATIONALE FOR INCLUSION IN REPORT	SOURCE
Table G-11	Brownfields	Federal and state lists of Brownfield sites (i.e., abandoned, idle, or underused commercial or industrial properties, where expansion or redevelopment is hindered by real or perceived contamination). EPA and Ecology maintain separate lists of Brownfields sites.	Sites with toxic chemicals present in various environmental media in the vicinity of the LDW.	EDR (2006b) from EPA and Ecology databases
Table G-12	Toxics Release Inventory System	TRIS facilities are registered with EPA and identified as facilities that release toxic chemicals to the air, water, and land in reportable quantities under the Superfund Amendments and Reauthorization Act, Title III, Section 313.	Facilities with toxic chemical releases to various environmental media in the LDW surface drainage area.	EDR (2006b) from EPA database
Table G-13	PSCAA's Registered Air Emission Sources	A list of sites registered with the PSCAA as producing emissions of air pollutants.	Facilities with registered air emissions in the vicinity of the LDW.	PSCAA (Nehen 2007)
Table G-14	Ecology Spills Program[a]	Spills reported to Ecology's Environmental Report Tracking System.	Spills in the LDW drainage basin.	EDR (2006b) from Ecology database
Table G-15	Seattle Public Utilities Spill Response Program Records[a]	SPU maintains a record of hazardous material spills involving the city's infrastructure.	Spills in the LDW drainage basin.	Schmoyer (2007b)

[a] Databases include only a partial list of spill information maintained by the USCG.

CSCSL – Confirmed and Suspected Contaminated Sites List
CSO – combined sewer outfall
Ecology – Washington State Department of Ecology
EDR – Environmental Data Resources, Inc.
EPA – US Environmental Protection Agency
ERTS – Environmental Report Tracking System
LDW – Lower Duwamish Waterway
LUST – leaking underground storage tank
NPDES – National Pollutant Discharge Elimination System
PSCAA – Puget Sound Clean Air Agency
RCRA – Resource Conservation and Recovery Act
SARA – Superfund Amendments and Reauthorization Act
SD – storm drain
SPU – Seattle Public Utilities
SWF/LF – solid waste facility/landfill
TRIS – Toxics Release Inventory System
USCG – US Coast Guard
UST – underground storage tank

Table 9-4. Databases used to identify facility information on potential sources to the LDW but excluded from Appendix G

DATABASE	DESCRIPTION	RATIONALE FOR EXCLUSION FROM THE REPORT	SOURCE
Comprehensive Environmental Response, Compensation, and Liability Information System	EPA-maintained database that contains information on potentially hazardous waste sites as reported by states, local municipalities, and private entities under Section 103 of CERCLA.	Facilities with toxic chemical releases to various environmental media in the vicinity of the LDW. The seven facilities identified on this database are either included previously on other database listings or are outside the LDW drainage basin.	EDR (2006b) from EPA database
Resource Conservation and Recovery Act – treatment, storage, and disposal facilities	EPA-maintained database that provides selective information on sites that store, transport, treat, or dispose of hazardous materials as defined by RCRA.	No RCRA-TSD sites were identified within the LDW drainage basin.	EDR (2006b) from EPA database
National Priorities List	EPA's list of Superfund sites is a subset of CERCLIS that identifies priority sites for cleanup under the Superfund program.	The NPL list is a subset of CERCLIS. The three NPL listed sites included the LDW, Harbor Island, and Pacific Sound Resources (located to the west of the mouth of the West Waterway)	EDR (2006b) from EPA database
Corrective Action Sites	Database maintained by EPA that tracks hazardous waste handlers with RCRA corrective action activity.	Entries in the CORRACTS database primarily include administrative compliance issues and do not have direct implications for source control.	EDR (2006b) from EPA database
RCRA small-quantity waste generators	EPA database that tracks facilities producing between 100 and 1,000 kg of hazardous waste per month (small-quantity waste generators).	Over 1,000 facilities within the EDR search radius were identified as being RCRA SQGs. Operations and chemical use/storage at facilities with SQG classifications vary greatly; an SQG listing provides little information relevant to potential LDW source control concerns. Therefore, tabularized summation of the SQG information was not completed. Other databases were relied upon to identify potential sources of contamination. In addition, SPU business inspections, which are discussed in this section, have focused on a more in-depth assessment of small-quantity waste generators in the LDW.	EDR (2006b) from EPA database
Emergency Response Notification System	Database maintained by the National Response Center, US Coast Guard to track reported releases of oil and hazardous substances.	Ecology and SPU databases were relied upon to identify hazardous materials spills information.	EDR(2006b) from National Response Center database
Hazardous Sites List	Database maintained by Ecology listing a subset of CSCSL sites ranked by the Washington Ranking Method.	The HSL is a subset of the CSCSL; CSCSL sites within the summation area are summarized in Table G-6.	EDR (2006b) from Ecology database
Voluntary Cleanup Program sites	List of sites maintained by Ecology that tracks sites that have entered either the VCP or the Independent Remedial Action Program.	Sites listed in the VCP database that are still undergoing cleanup or investigation are also listed on the CSCSL, dry cleaners, or LUST databases, all of which are summarized in Appendix G.	EDR (2006b) from Ecology database

CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
CERCLIS – Comprehensive Environmental Response, Compensation, and Liability Information System
CORRACTS – Corrective Action Sites
CSCSL – Confirmed and Suspected Contaminated Sites List
Ecology – Washington State Department of Ecology
EDR – Environmental Data Resources, Inc.
EPA – US Environmental Protection Agency
ERNS – Emergency Response Notification System
HSL – Hazardous Sites List
IRA – independent remedial action
LDW – Lower Duwamish Waterway
LQG – large-quantity generator
LUST – leaking underground storage tank
NPL – National Priorities List
RCRA – Resource Conservation and Recovery Act
SPU – Seattle Public Utilities
SQG – small-quantity generator
TSD – treatment, storage, and disposal
USCG – US Coast Guard
VCP – Voluntary Cleanup Program
WARM – Washington Ranking Method

Appendix G consists of lists of sites from various databases as provided in agency files. These lists are intended to provide a general indication of the scope and complexity of the source control environment of the LDW, and not to identify specific sources. The inclusion or exclusion of a facility on one of these lists does not infer that it is or is not a potential source of contamination to the LDW. In addition, none of the resource information has been independently verified beyond the citations noted, and LDWG cannot verify the completeness or the accuracy of the information.

Furthermore, much of the primary data have had varying levels of quality control and data validation, which have not generally been documented in the resource materials reviewed. Therefore, information included in Appendix G should not be relied upon as complete and factual. Additional evaluations of the data would be required in order to use the data quantitatively or as an inference to identify potential sources. In many cases, the most recent database information available for a facility is considered outdated, and likely does not represent the current status or condition of the site.

In order to focus the tabularized summation of the EDR presentation, the following geographic parameters were used to identify which property reviews were most relevant to source control in the LDW:

- Facilities located within the LDW surface drainage basin, as defined by King County and the City of Seattle (King County and SPU 2005)
- Facilities within 1 mi of the shoreline of the LDW study area
- Facilities within 1 mi upriver of the Upper Turning Basin

Facilities meeting these criteria and included in the EDR documentation are summarized in tables in Appendix G; Tables 9-3 and 9-4 summarize the types of information in each environmental database, their relevance to source control in the LDW, and the rationale for their inclusion. Listed facilities identified include those on Ecology's Confirmed and Suspected Contaminated Sites List (CSCSL), RCRA- and CERCLA-listed properties, registered Brownfield properties, and leaking underground storage tank (LUST) sites. In addition to the databases provided by EDR, business inspection programs conducted by King County and Seattle Public Utilities (SPU) also provided information about potential sources; these programs are described in Section 9.4.4.5.

9.4.1.3 Agency and Source Control Work Group information

The final source of information used to evaluate potential ongoing sources to the LDW included work products and personal communications from Ecology, the City of Seattle, King County, the Port of Seattle, PSCAA, USACE, and EPA. Information on upland sites is available in existing SCAPs, data gaps reports, property reviews, and other regulatory information sources (such as the databases listed in Table 9-3), along with direct discussions with Ecology, SCWG, and EPA representatives.

Other readily available information used to summarize the status of source control in the LDW included sediment sampling in CSO/SD systems and road rights-of-way (ROWs) conducted by the City (in conjunction with Boeing at some locations), business inspections conducted by the City and County, a street sweeping pilot program conducted by the City, atmospheric deposition and water quality sampling conducted by the County, air quality information provided by PSCAA and Ecology, and information on the sources, fate, and transport of phthalates gathered by the Sediment Phthalates Work Group (SPWG). Readily available information was reviewed, and agency personnel were contacted for additional information when necessary.

9.4.2 Atmospheric loading and assessment programs

Common air pollutants with the ongoing potential to affect LDW sediments through deposition include metals, PCBs, PAHs, dioxins and furans, and phthalates (EPA 2001; Floyd|Snider 2007a). PSCAA records show that over 200 businesses in the Duwamish Valley are registered as active sources of air pollution (listed in Appendix G) (PSCAA 2007a). Motor vehicle traffic on Interstate 5 (I-5) and local roads also produces nitrous oxide, black carbon, and other emissions through the burning of fossil fuels (PSCAA 2006a).

Air quality and atmospheric deposition information has been collected in the vicinity of the LDW by several groups, including the PSCAA, Ecology, and King County. Efforts are also underway to research the sources, transport, and fate of air pollutants and to reduce emissions through regulations and voluntary programs. Air emissions monitoring and assessment programs are discussed in Section 9.4.2.2, and emission reduction programs and additional regulations are discussed in Section 9.4.2.5.

9.4.2.1 Particulate matter

PM refers to solid particulates and liquid droplets suspended in the air. There are two size classes of PM: fine particulate matter (FPM), which refers to particles less than 2.5 µm in diameter ($PM_{2.5}$), and coarse particulate matter (CPM), which includes particles between 2.5 and 10 µm in diameter (PM_{10}) (EPA 2008a). Fine PM (less than 2.5 µm in diameter) is of primary concern in the Puget Sound region (PSCAA 2006a). Particles greater than 10 µm in diameter can be generated by grinding activities and through dust re-suspension. Although particles of this size generally do not travel great distances, they can be a potential source of some metals (e.g., copper and zinc) and other chemicals in localized areas, such as along high-traffic roadways.

FPM is generated through the burning of wood, fossil fuels, and other materials and can also be formed when pollutant gases react in the atmosphere (King County and SPU 2005; PSCAA 2006d). In the Puget Sound region, FPM is the air pollutant of greatest concern for human health (PSCAA 2006d); it can also provide a pathway for the transport of chemicals, such as metals, PCBs, dioxins and furans, phthalates, and cPAHs (Floyd|Snider 2007a, b; EPA 2001; King County and SPU 2005). At some stations, atmospheric deposition flux data for some PAHs collected by King County also showed a correlation with FPM concentration data in the atmosphere (King County 2008b). FPM has the potential to affect sediment quality in the LDW if chemicals released to the atmosphere attach to the fine particles that are then deposited through direct or indirect atmospheric deposition to the LDW. Once in the LDW, FPM and associated chemicals have the potential to accumulate in bottom sediments. The 3-year average 98th percentile daily concentrations of FPM in the Duwamish area were approximately 34 µg/m^3 (as reported in 2001) and 29 µg/m^3 (as reported in 2005) (PSCAA 2006a).

9.4.2.2 Toxic air pollutant monitoring and assessment

Toxic air pollutants, as classified by EPA, are those that have the potential to adversely affect human health (EPA 2007a). They include PAHs, PCBs, phthalates, dioxins and furans, and several metals. Common emission sources of toxic air pollutants include industrial facilities, chemical manufacturing plants, solid waste incinerators, motor vehicles, marine vessels, trains, and wood smoke (PSCAA 2006b, c). Use of solvents, paints, degreasers, and other chemicals can also release toxic air pollutants (EPA 2007a). Ecology monitors concentrations of these pollutants at Beacon Hill in Seattle. This is done as part of EPA's national air toxics trends site network. Other temporary (1-year) monitoring studies have been conducted in Seattle, Tacoma, Spokane, and Longview. These studies characterize air emissions and identify toxic air pollutants that could pose risks to human health (PSCAA 2003). These pollutants have the potential to affect sediment quality if they migrate to the LDW through direct or indirect atmospheric deposition.

WSDOH conducted a health consultation to assess potential health risks from air pollution in the South Park and Georgetown neighborhoods of South Seattle (WSDOH 2008b). The study used a model to estimate risks to human health from air emissions from various sources, and several estimates were made using the model. Emission sources included in the model were point sources (facilities registered with PSCAA), mobile sources including motor vehicles, and wood stove emissions. Emissions from railways, aircraft, and ships were not included in the inventory because estimates of these emissions were not available.

The study concluded that on-road mobile sources (e.g., cars and trucks) posed the greatest health risks from air pollution in South Seattle. Seventy-four percent of the theoretical cancer risk from on-road sources was attributed to diesel PM (which consists of both CPM and FPM). Point sources were also found to contribute to health risks, primarily because of chromium compound emissions. Wood stove and fireplace use was also found to contribute to health risks from air pollution. The Puget Sound Air Toxics Evaluation (PSCAA 2003) also concluded that the majority of cancer risk associated with air pollutants comes from diesel exhaust. The study used both Ecology monitoring data and an analysis and comparison with modeling results from EPA's national air toxics assessment.

King County has collected two phases of atmospheric deposition data in the LDW basin to provide information for source control. Four rounds of data were collected in Phase 1 of sampling, which was conducted between January and May 2005.[145] Deposition flux values were calculated for phthalates and PAHs based on these data. Sixteen rounds of data were collected in Phase 2 of sampling, which was conducted between October 2005 and April 2007. Deposition flux values were calculated for PCB Aroclors in addition to phthalates and PAHs based on the Phase 2 data. No dioxin and furan monitoring has been conducted as part of this program (King County 2008b).

Four stations were used during Phase 1 monitoring: Beacon Hill (BW), Duwamish Valley/E Marginal Way S (CE), Georgetown (DZ), South Park Community Center (SPCC) (Map 9-7) (King County 2008b). In Phase 2, (KCIA was added, and Beacon Hill and Duwamish Valley stations were relocated (BWR and CER, respectively) to areas thought to be more representative of neighborhood-scale atmospheric conditions. The Duwamish (CE/CER), Georgetown (DZ), and South Park Community Center (SPCC) stations are most representative of the Duwamish industrial basin, the Beacon Hill (BW/BWR) station is most representative of Seattle urban-scale conditions, and the KCIA station represents a location impacted by mobile sources (fuel combustion from aircraft). Atmospheric deposition samples were collected using a passive atmospheric deposition sampler that samples for "dry" and "wet" deposition combined. Atmospheric deposition flux values indicating mass deposition/area/time were provided for each chemical. The minimum, maximum, mean, median, 25th percentile, and 75th percentile results by chemical are provided for the Duwamish industrial basin stations (CE/CER, DZ, and SPCC), the Seattle urban-scale station (BW/BWR), and the airport station (KCIA) (Table 9-5).

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[145] The Phase 1 atmospheric deposition sampling was conducted primarily as a pilot study; the sampler was redesigned for the Phase 2 study. Data presented in this section are from the Phase 2 study.

Table 9-5. Summary results for King County's Phase 2 atmospheric deposition data

CHEMICAL	SAMPLE COUNT	COUNT OF DETECTED VALUES	DETECTION FREQUENCY (%)	DETECTED DEPOSITION FLUX VALUES (µg/m^2/day)						DEPOSITION FLUX VALUES WITH ND = 0[a]		
				MIN	MAX	MEAN	MEDIAN	25TH PERCENTILE	75TH PERCENTILE	MIN	MAX	MEAN
Duwamish Industrial Basin Stations (CE/CER, DZ, SPCC)												
PAHs												
Benzo(a)anthracene	47	38	81	0.003	0.243	0.063	0.051	0.024	0.077	0	0.243	0.051
Benzo(a)pyrene	47	29	62	0.008	0.265	0.095	0.072	0.052	0.135	0	0.265	0.059
Benzo(b)fluoranthene	47	40	85	0.010	0.317	0.119	0.103	0.082	0.154	0	0.317	0.102
Benzo(g,h,i)perylene	47	40	85	0.010	0.323	0.121	0.104	0.062	0.155	0	0.323	0.103
Benzo(k)fluoranthene	47	36	77	0.009	0.317	0.101	0.084	0.057	0.131	0	0.317	0.077
Chrysene	47	46	98	0.037	0.464	0.146	0.123	0.093	0.178	0	0.464	0.142
Dibenzo(a,h)anthracene	47	17	36	0.020	0.170	0.042	0.030	0.022	0.043	0	0.170	0.015
Indeno(1,2,3-cd)pyrene	47	34	72	0.005	0.232	0.078	0.060	0.046	0.100	0	0.232	0.056
Pyrene	47	46	98	0.088	0.831	0.241	0.192	0.136	0.271	0	0.831	0.235
Phthalates												
BEHP	47	47	100	0.261	12.240	2.940	2.255	1.396	3.609	na	na	na
BBP	47	47	100	0.163	7.007	0.950	0.599	0.299	1.063	na	na	na
Diethyl phthalate	47	42	89	0.007	0.447	0.165	0.142	0.076	0.217	0	0.447	0.147
Dimethyl phthalate	47	34	72	0.029	0.153	0.068	0.061	0.049	0.080	0	0.153	0.049
Di-n-butyl phthalate	47	39	83	0.002	0.678	0.177	0.093	0.034	0.249	0	0.678	0.147
Di-n-octyl phthalate	47	27	57	0.037	2.874	0.658	0.329	0.159	0.748	0	2.874	0.378
PCBs												
Total PCBs	30	6	20	0.014	0.065	0.039	0.042	0.019	0.039	0	0.065	0.008
Seattle Urban-Scale Stations (BW/BWR)												
PAHs												
Benzo(a)anthracene	12	6	50	0.019	0.030	0.023	0.022	0.021	0.023	0	0.030	0.011
Benzo(a)pyrene	12	5	42	0.021	0.026	0.023	0.021	0.021	0.025	0	0.026	0.009

Table 9-5, cont. Summary results for King County's Phase 2 atmospheric deposition data

Chemical	Sample Count	Count of Detected Values	Detection Frequency (%)	Detected Deposition Flux Values ($\mu g/m^2/day$)						Deposition Flux Values With ND = 0[a]		
				Min	Max	Mean	Median	25th percentile	75th percentile	Min	Max	Mean
Benzo(b)fluoranthene	12	10	83	0.033	0.072	0.047	0.046	0.038	0.049	0	0.072	0.039
Benzo(g,h,i)perylene	12	8	67	0.033	0.070	0.047	0.043	0.038	0.057	0	0.070	0.031
Benzo(k)fluoranthene	12	7	58	0.025	0.054	0.037	0.035	0.031	0.040	0	0.054	0.021
Chrysene	12	12	100	0.023	0.090	0.055	0.051	0.038	0.074	na	na	na
Dibenzo(a,h)anthracene	12	1	8	0.012	0.012	na	na	na	na	0	0.012	0
Indeno(1,2,3-cd)pyrene	12	5	42	0.025	0.059	0.034	0.027	0.025	0.033	0	0.059	0.014
Pyrene	12	12	100	0.035	0.157	0.090	0.086	0.073	0.110	na	na	na
Phthalates												
BEHP	12	12	100	0.955	3.479	1.640	1.457	1.151	1.993	na	na	na
BBP	12	12	100	0.193	0.980	0.498	0.495	0.271	0.686	na	na	na
Diethyl phthalate	12	11	92	0.092	0.658	0.209	0.135	0.094	0.276	0	0.658	0.191
Dimethyl phthalate	12	11	92	0.022	0.104	0.039	0.034	0.028	0.035	0	0.104	0.035
Di-n-butyl phthalate	12	7	58	0.015	0.143	0.081	0.070	0.037	0.130	0	0.143	0.047
Di-n-octyl phthalate	12	5	42	0.096	0.165	0.128	0.137	0.099	0.144	0	0.165	0.053
PCBs												
Total PCBs	8	0	0	nd	nd	nd	nd	na	na	0	0	0
KCIA												
PAHs												
Benzo(a)anthracene	16	16	100	0.185	1.473	0.928	0.976	0.679	1.180	na	na	na
Benzo(a)pyrene	16	16	100	0.234	2.225	1.324	1.419	0.947	1.691	na	na	na
Benzo(b)fluoranthene	16	16	100	0.375	3.410	2.082	2.123	1.619	2.656	na	na	na
Benzo(g,h,i)perylene	16	16	100	0.259	2.494	1.373	1.517	0.983	1.591	na	na	na
Benzo(k)fluoranthene	16	16	100	0.310	2.774	1.596	1.664	1.248	1.976	na	na	na
Chrysene	16	16	100	0.347	3.078	1.958	2.063	1.474	2.590	na	na	na
Dibenzo(a,h)anthracene	16	16	100	0.072	0.579	0.391	0.418	0.305	0.507	na	na	na
Indeno(1,2,3-cd)pyrene	16	16	100	0.228	2.149	1.228	1.312	0.879	1.521	na	na	na

Table 9-5, cont. Summary results for King County's Phase 2 atmospheric deposition data

CHEMICAL	SAMPLE COUNT	COUNT OF DETECTED VALUES	DETECTION FREQUENCY (%)	DETECTED DEPOSITION FLUX VALUES (µg/m²/day)						DEPOSITION FLUX VALUES WITH ND = 0[a]		
				MIN	MAX	MEAN	MEDIAN	25TH PERCENTILE	75TH PERCENTILE	MIN	MAX	MEAN
Pyrene	16	16	100	0.574	4.652	2.712	2.814	1.891	3.440	na	na	na
Phthalates												
BEHP	16	16	100	0.268	6.144	1.719	1.512	1.159	1.941	na	na	na
BBP	16	16	100	0.187	2.913	0.843	0.585	0.257	0.922	na	na	na
Diethyl phthalate	16	15	94	0.026	0.472	0.236	0.217	0.128	0.355	0	0.472	0.221
Dimethyl phthalate	16	11	69	0.031	0.100	0.053	0.052	0.040	0.059	0	0.100	0.036
Di-n-butyl phthalate	16	14	88	0.033	0.630	0.211	0.109	0.054	0.321	0	0.630	0.185
Di-n-octyl phthalate	16	7	44	0.057	0.651	0.193	0.125	0.094	0.166	0	0.651	0.084
PCBs												
Total PCBs	10	2	20	0.043	0.045	0.044	0.044	na	na	0	0.045	0.009

Source: (King County 2008b)

[a] Calculations were performed with only detected values.

BEHP – bis(2-ethylhexyl) phthalate
BBP – butyl benzyl phthalate
KCIA – King County International Airport
na – not applicable
nd – not detected
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl

During the Phase 2 monitoring, the detection frequencies for individual PAHs ranged from 36% to 98% at the Duwamish industrial area stations, and from 8% to 100% at the Beacon Hill station, which represents urban neighborhood conditions (Table 9-5). All individual PAHs analyzed were detected in 100% of the samples from the KCIA station. The prevalence of PAHs at the KCIA station is attributed to mobile sources (i.e., aircraft) (King County 2008b). Mobile combustion sources (i.e., cars and trucks) are also thought to be important sources of PAHs at the Duwamish industrial area stations. Pyrene had the highest concentration at all three monitoring station groups. The highest pyrene flux was 0.831 µg/m^2/day at the Duwamish industrial area stations, 0.157 µg/m^2/day at the Beacon Hill station, and 4.652 µg/m^2/day at the KCIA station.

The ranges of detection frequencies for phthalates were similar at all three monitoring station groups, although the minimum detection frequency was highest for the Duwamish industrial area stations (Table 9-5). BEHP and BBP were detected in 100% of samples from all three monitoring station groups. The highest BEHP and BBP fluxes were detected at the Duwamish industrial area stations (12.24 µg/m^2/day and 7.007 µg/m^2/day, respectively).

PCBs were detected in 20% of the samples collected from the Duwamish industrial area stations and in 20% of the samples collected from the KCIA station. PCBs were not detected in any of the samples collected from the Beacon Hill station, which represents urban neighborhood conditions. MDLs for PCBs corresponded to deposition flux values that ranged from 0.011 to 0.063 µg/m^2/day. The highest detected total PCB atmospheric deposition flux result was 0.065 µg/m^2/day and the lowest detected total PCB atmospheric deposition flux result was 0.014 µg/m^2/day (both detected at the Duwamish industrial area stations).

The LDW atmospheric deposition flux data collected by King County were compared with data collected by EPA and Ecology from stations near the LDW, and to data collected from other areas (the Georgia Basin of British Columbia, Denmark, the greater Puget Sound area, and the Great Lakes region) by other entities (King County 2008b). A comparison of deposition flux values among studies for BBP, BEHP, PAHs, and total PCBs can be made using average values from each study (Table 9-6). In general, the results from the LDW appear to be similar to those from other urban environments; however, some differences were noted (Table 9-6). Differences in LDW deposition flux values and those from other studies are likely the result of variations in sampling methods and intervals, among other study-specific factors (King County 2008b).

Table 9-6. Average atmospheric deposition flux values for the LDW compared with those from other studies

Study or Program	Study Location	Types of Deposition Measured in Study[a]	Average Atmospheric Deposition Flux Value ($\mu g/m^2/day$)			
			BBP	BEHP	PAHs	Total PCBs
King County Lower Duwamish Waterway Passive Atmospheric Deposition Sampling	LDW (Duwamish industrial basin stations)[b, c]	wet and dry deposition	0.95[d, e]	2.94 [d, e]	0.24 (pyrene) 0.1 (benzo(a)pyrene) 0.11 (all PAHs included in study)[d, e, f]	0.037[d, e]
Environment Canada Georgia Basin/ Puget Sound Airshed study	Georgia Basin (Chilliwack station, mixed rural/urban land use)	dry deposition	0.063[g]	0.615[g]	0.063[d, g]	na
Phthalates and Nonylphenols in Roskilde Fjord	Roskilde Fjord	deposition type not reported	0.047	0.625	na	na
Washington State Department of Ecology Air Quality Program	Puget Sound Region (Beacon Hill station)[c]	dry deposition and gas phase absorption	na	na	0.207 (pyrene)[g]	na
	Puget Sound Region (Georgetown station)[c]				0.346 (pyrene)[g]	
US. EPA/ Environment Canada Integrated Atmospheric Deposition Network	Great Lakes Region (IIT/Chicago station, urban land use)	wet and dry deposition and gas phase absorption	na	na	0.085 – 0.261[g] (benzo(a)pyrene)	0.128 – 0.654[h]
New Jersey Atmospheric Deposition Network	New Jersey (Camden station, urban/industrial land use)	wet and dry deposition	na	na	0.154 (pyrene) 0.069 (benzo(a)pyrene)	0.096
	New Jersey (Jersey City station, urban/industrial land use)		na	na	0.270 (pyrene) 0.113 (benzo(a)pyrene)	0.034

Sources: King County and SPU (2005) and King County (2008b)

[a] Data from different studies are directly comparable only when the same types of atmospheric deposition were measured. The LDW passive atmospheric deposition study sampled wet and dry deposition. Some of the comparison studies did not include wet deposition, and others included measures of gas phase absorption in addition to wet and/or dry deposition. Different chemicals are deposited through different modes of deposition. As an example, studies that include gas phase sampling measure lower molecular weight PCB congeners, which are primarily deposited through gas phase absorption. Results for total PCBs from studies that include gas phase absorption measurements will generally have higher total PCB results because of the addition of lower molecular weight PCB congeners.

[b] The Duwamish industrial basin stations include the Duwamish station (CE/CER), the Georgetown station (DZ), and the South Park Community Center station (SPCC).

[c] King County and Ecology used some of the same monitoring stations (including the Georgetown and Beacon Hill stations) in their studies.

[d] Average calculations were made using only Phase 2 data because the deposition sampler design was changed between the Phase 1 and Phase 2 sampling events.

[e] Average calculations include only detected deposition flux values.

[f] Value presented for all PAHs is an average of the average deposition flux values provided for seven individual cPAHs and two additional individual PAHs (benzo(g,h,i)perylene and pyrene).

[g] This study presented results as air concentrations; the average deposition flux values were calculated using an atmospheric deposition velocity value of 0.2 cm/second.

[h] Represents a range of averages.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
EPA – US Environmental Protection Agency
IIT – Illinois Institute of Technology
LDW – Lower Duwamish Waterway
na – not available
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl

Differences in study methodology also introduce uncertainty in the comparison of LDW flux values to flux values from other studies (King County 2008b). Because the atmospheric deposition monitoring conducted in the LDW did not include gas absorption, some chemicals such as LPAHs may have been underestimated. Data collected by Environment Canada, Ecology, and EPA/ Environment Canada Integrated Atmospheric Deposition Network were reported as average air concentration values and were converted to average atmospheric deposition flux values using a deposition velocity value of 0.2 cm/s. This conversion introduces uncertainty into the comparison of values from various studies.

Results of the atmospheric deposition monitoring for the LDW basin indicate atmospheric deposition may contribute phthalates, PAHs, and PCBs to the LDW; however, the relative contribution of atmospheric deposition compared with other sources was not evaluated for the LDW. A comparison of chemical loadings from direct atmospheric deposition to loadings from other pathways was made in the study of toxic chemical loadings to Puget Sound conducted in 2007 (Hart Crowser et al.). This study used the atmospheric deposition data collected in the LDW by King County as well and data from other studies. Information provided in this study can be applied to the LDW for a general understanding of the potential for atmospheric deposition to contribute toxic pollutants to LDW sediments.

The study of toxic chemical loadings to Puget Sound reported that direct atmospheric deposition appears to be a significant pathway to Puget Sound for some chemicals (Hart Crowser et al. 2007). Loading estimates for direct atmospheric deposition were compared with loading estimates for surface water runoff; surface water runoff was identified as the pathway that transported the majority of toxic chemicals to Puget Sound. For PAHs, direct atmospheric deposition represented an equal or greater loading source than surface water runoff. For other chemicals, such as BEHP, DDT, dioxins and furans, and PCBs, direct atmospheric deposition represented a small percent (approximately 4 to 7%) of the loading estimate for the surface water runoff pathway. For metals, loading estimates from atmospheric deposition were between 5 and 35% of estimates for surface water runoff. The report acknowledged the uncertainty associated with these loading estimates because of limited atmospheric deposition data (Hart Crowser et al. 2007).

Gas absorption may be the primary mechanism for atmospheric deposition of SVOCs, including LPAHs (Hart Crowser et al. 2007). No gas absorption rates were available for the SVOCs identified as chemicals of concern to Puget Sound. Therefore, estimates of LPAH loading via this pathway likely had a high degree of uncertainty.

King County currently has no plans for additional monitoring of atmospheric deposition in the Duwamish Valley (Tiffany 2007). The PSCAA, in coordination with Ecology and under a grant from EPA, will conduct an air toxics monitoring study at a monitoring station in the Duwamish Valley and at three stations in Tacoma (Himes 2008). Results from the study will be available in 2010. The sampling program will provide information on spatial and temporal trends in air toxics concentrations, and provide data for comparison of air toxics concentrations between cities (Seattle and Tacoma) and between local areas and national trends.

Ecology also has plans to continue air monitoring at the Beacon Hill station under the National Air Toxics program, and to conduct a deposition study in the Puget Sound in association with the Puget Sound Partnership (Williamson 2008). The deposition study began in the fall of 2008; data will be collected for approximately one year. Monitoring stations are located throughout the Puget Sound region, and all monitors are located in shoreline areas. The study is using similar sampling methods to the King County study.

9.4.2.3 Sediment Phthalates Work Group

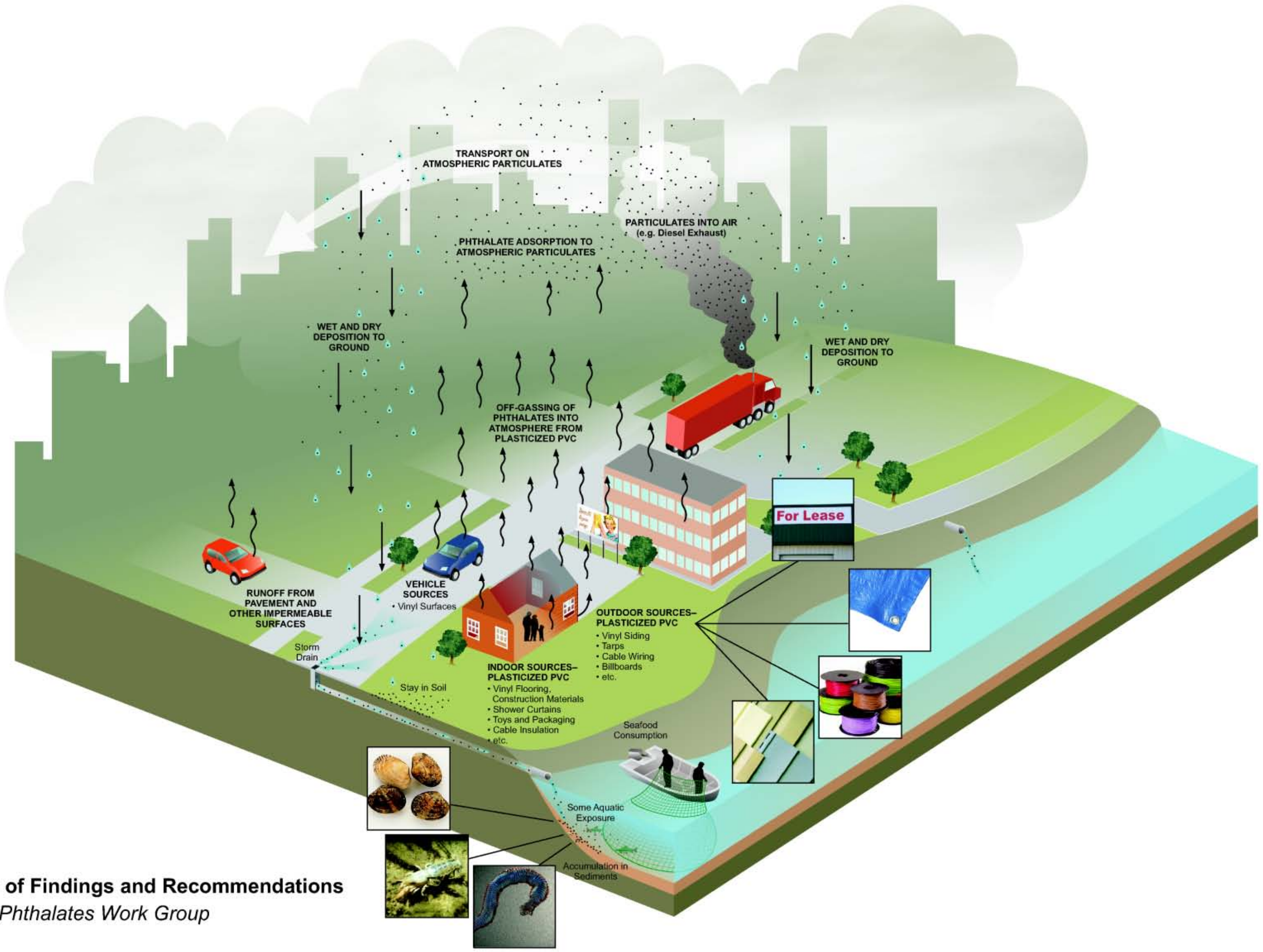
The SPWG was a voluntary work group that included representatives from the City of Seattle, the City of Tacoma, King County, Ecology, and EPA (Ecology 2007m). The categories of information the SPWG studied were phthalate occurrence, phthalate risk and receptors, source identification, source control and treatment, and federal and state regulations relevant to phthalate contamination in sediments (Floyd|Snider 2007a). As stated in the Letter of Agreement for the work group, the goals of the SPWG were to "work together to summarize and evaluate existing information regarding phthalate sediment contamination issues, identify data gaps and provide recommendations on how to address these phthalate sediment contamination dilemmas in the short-term" and to "make recommendations for further study and describe possible frameworks to address sediment phthalate issues for the longer term" (Sediment Phthalates Work Group 2006). An additional goal of the SPWG's was to provide recommendations for preventing phthalate recontamination of the LDW after cleanup (Floyd|Snider 2007a).

The SPWG developed a basic working model of how phthalates from non-point sources may reach the LDW sediments (Floyd|Snider 2007b). This model has not been peer-reviewed or otherwise validated. The basic premise is that phthalates initially enter the environment primarily through off-gassing from manufactured products. Once in the atmosphere, they attach to PM and are distributed to land and water surfaces. They are transported to water bodies like the LDW through stormwater runoff. Numerous sources of phthalates exist. An earlier phthalate source sampling project conducted in 2003 identified some common products that contain phthalates (Section 9.4.2.4). The 2003 study of possible phthalate-bearing material was not comprehensive or definitive; it was simply aimed at identifying common materials with high concentrations of phthalates in hopes of finding easily controlled sources to sediments.

The SPWG concluded that phthalates are widespread in urban and other developed environments and that they are ubiquitous in water, soil, sediment and air (Floyd|Snider 2007a). Figure 9-3 presents a visual summary of the SPWG's conceptual model of how phthalates move through the environment from various sources into sediments. This figure was presented with the SPWG's recommendations and findings (Floyd|Snider 2007a).

Phthalates from non-point sources can accumulate in sediment through both direct and indirect atmospheric deposition, and atmospheric deposition has been identified as the pathway most likely contributing the majority of phthalates to sediments (Floyd|Snider 2007a). However, point sources of phthalates (i.e., direct discharges from industrial processes or wastes) may also exist. Based on its studies, the SPWG did not identify any single, large, easily controllable source of phthalates to sediments, although a few such sources have been identified in the past. For example, a facility called Janco-United, Inc., formerly located in the 4400 block of 4th Avenue S, disposed of degreasing compounds and other wastes associated with janitorial supply operations that allegedly contained phthalates. These wastes were initially discharged directly to the ground and then through an SD that discharged to the LDW; the discharges occurred for several years in the early 1980s.

The two phthalates most frequently detected in Puget Sound sediments, BEHP and BBP, are commonly attached to PM in the air. When indirect atmospheric deposition occurs, phthalates deposited on surfaces throughout an entire watershed can be carried through surface water run-off (including CSO and SD discharges) to discrete sediment areas where they then accumulate (Floyd|Snider 2007a). Although atmospheric deposition and surface water runoff are the pathways by which phthalates reach sediments, the concentrations of phthalates in these media do not generally trigger regulatory responses based on human or ecological health risks. In sediment, however, phthalates can accumulate at concentrations above sediment cleanup standards.



Summary of Findings and Recommendations
Sediment Phthalates Work Group

Source: Floyd|Snider (2007b)

Figure 9-3. Sediment Phthalates Work Group conceptual model

Build-up of phthalates in sediment can be toxic to benthic invertebrates; organisms at higher trophic levels (i.e., humans) metabolize phthalates and excrete them and do not appear to be at risk (Floyd|Snider 2007b). The SPWG concluded that phthalates in sediments presented little additional human health risk compared with other chemicals and so recommended that sediment cleanup actions not be delayed because of the potential for phthalate recontamination. The group suggested that phthalates continue to be addressed through long-term operation and maintenance programs (Floyd|Snider 2007b). Methods for reducing and controlling sources of phthalates include using substitutes for phthalates in plasticized goods, removing particulates from the atmosphere, and reducing stormwater flow runoff by increasing pervious surfaces within watersheds.

As stated in the SPWG summary of findings and recommendations report (Floyd|Snider 2007b):

> "During the 10-month process, the Work Group collected, summarized, and evaluated existing information to:
>
> - Document where phthalates are found and identify potential sources
> - Define phthalate contamination concerns within current regulations
> - Place sediment phthalate concerns in perspective with other sediment contamination risks and within the broader issue of phthalate risks from other exposure pathways
> - Identify potential source control and treatment options
> - Provide recommendations on next steps
> - Share findings with the public"

The SPWG made several recommendations (Floyd|Snider 2007b), including:

- Phthalate recontamination at sediment cleanup sites should be managed with site-specific operation and maintenance plans.
- Other contaminants that are transferred through the air to stormwater to sediment pathway should be studied in order to further validate the SPWG's understanding of this pathway and to reduce additional contaminants (beyond phthalates) that reach sediments through this pathway.

9.4.2.4 Phthalate source studies

Atmospheric deposition is one likely source of phthalate compounds to the LDW. In 2003, as part of their compliance with their CERCLA AOC for the Thea Foss/Wheeler-Osgood Waterway, the City of Tacoma conducted a phthalate source study to investigate whether phthalates in stormwater solids varied with land use and to investigate particular materials (e.g., automotive fluids, tires, and packing peanuts) for phthalates. King County and the City of Seattle joined the effort to investigate materials for phthalates, and each entity looked at a separate set of materials for their phthalate content. The City of Tacoma analyzed automotive fluids and other automotive products and packing peanuts. The City of Seattle analyzed tires, belts, and roadway materials. King County conducted a study of air particulates and re-analyzed some of the materials the City of Tacoma had analyzed earlier (e.g., different types of fluids and brands of products). King County and City of Tacoma laboratories found high concentrations of BEHP in some automotive belts, brake pads, and tires (King County and SPU 2005). Liquid products generally contained lower concentrations of BEHP. Analytical results from product testing were reported in the City/County source control progress report from 2004 (King County and SPU 2004). The literature review suggested that some vehicle fuel products, such as diesel, contain BEHP that may be released into the atmosphere in exhaust (King County and SPU 2005, citing California Air Resources Board, 1997).

Dust samples were also collected during two sampling rounds from the roof of the Tacoma Dome as part of the phthalate source studies. The first round of sampling was conducted in May 2003 prior to roof washing, and the second round of sampling was conducted in July 2003 immediately after the roof was washed. BEHP was detected at 600 $\mu g/ft^2$ in the first round sample, and at 42 $\mu g/ft^2$ in the second round sample (King County and SPU 2004).

9.4.2.5 Air emissions reduction programs

EPA regulates air pollutants through the CAA (42 USC § 7401 et seq.), and local regulations include the Washington CAA (WAC 173-401) and PSCAA Regulations I, II, and III. PSCAA's regulations provide acceptable source impact levels (ASILs) for approximately 650 chemicals, which are used to evaluate the potential for a single emission source to impact air quality. PSCAA requires dischargers of toxic air pollutants to use best available control technology (BACT), described in their regulations, to minimize emissions of these chemicals and protect human health. Currently the Puget Sound region is in compliance with federal air quality standards (PSCAA 2007b). Except for FPM and ozone, concentrations of criteria air pollutants in Puget Sound have been in compliance with regulatory standards for nearly a decade (PSCAA 2006a).

Current regulations on air quality are generally based on protecting human health and may not be adequate to address source control concerns associated with atmospheric deposition. Several local groups have initiated emissions reduction programs that go beyond regulatory requirements to further control the release of atmospheric pollutants.

The Port of Seattle is helping to lead the Puget Sound Maritime Air Forum (PSMAF), an organization of private and public entities with regulatory responsibilities related to air emissions. The PSMAF is working to identify, quantify, and reduce maritime sources of air pollutants (PSMAF 2007). The primary goals of the forum are to ensure that the Georgia Basin-Puget Sound area continues to meet ambient air quality standards and to reduce port-related air quality impacts on human health and climate change (PSMAF 2007). Through the use of cleaner burning fuels, cleaner engines, and other technology, PSMAF is working to reduce FPM emitted by ships at berth by 70% by 2010. The group is also working to reduce emissions from trucks, trains, and other harbor craft used in Port activities. A reduction in fuel-burning emissions would result in a reduction in air pollutant loading, and a reduction in FPM could also potentially result in a reduction in transport of chemicals such as metals, phthalates, and PAHs to LDW sediments through direct and indirect atmospheric deposition.

The PSMAF has completed an air emissions inventory to identify air pollutants originating from diesel equipment used in maritime operations in the Puget Sound region (Starcrest Consulting Group 2007). PM and VOCs were two types of pollutants included in the inventory. The emissions inventory estimated that 1,270 tons per year (tpy) of VOCs are emitted from maritime sources, while 19,200 tpy are emitted from non-maritime sources; 220 tpy FPM are emitted through maritime emissions, while 3,600 tpy are emitted from non-maritime emissions; and 170 tpy diesel particulate matter (a type of FPM) are emitted through maritime emissions, while 260 tpy are emitted through non-maritime emissions (Starcrest Consulting Group 2007). Data provided in the inventory will be used to prioritize pollution reduction efforts.

The PSMAF is part of PSCAA's Diesel Solutions Program, which was initiated in 2001. The program works to reduce diesel emissions from numerous sources, including the construction industry, railways, and transit vehicles (PSCAA 2007b). Reducing diesel emissions will reduce FPM and can play a role in the reduction of other air pollutants that are transported through the atmosphere when they adsorb to FPM.

9.4.3 Surface water inputs

The Green/Duwamish River drainage area is approximately 1,466 km^2 (566 mi^2) and is part of WRIA 9 (Kerwin and Nelson 2000). According to information on WRIA 9, the Green/Duwamish River flows through four subwatersheds: the upper Green River, the middle Green River, the lower Green River, and the Duwamish Estuary (Map 2-2). The LDW is located within the Duwamish Estuary subwatershed. The flow of water to the LDW from upriver provides a potential source of chemicals to the study area. Upstream data are presented in Section 7.

Tributary creeks within the LDW drainage basin and direct surface water runoff are also potential sources of chemical inputs to the system. Two main creeks within the Duwamish River Estuary subwatershed, Puget Creek and Hamm Creek, drain into the LDW. The following sections discuss the potential upriver sources, tributary inputs, and direct surface water runoff to the LDW.

9.4.3.1 Upper Duwamish/Green River

Both historical and current land use practices in the Green/Duwamish River watershed provide the potential for pollutants to enter the river upstream of the LDW study area. These potential upstream sources include:

- Fecal coliform bacteria from human sewage discharges and animal wastes (both domestic and wild) (Herrera 2006)
- Pesticide and fertilizer runoff from agricultural operations
- Chemicals from various sources, including but not limited to roadways, industrial and commercial facilities, and surface water runoff from mining operations
- Chemicals deposited by atmospheric deposition
- Contaminated sediment from logging and sawmill operations and urban development throughout the watershed

Historically, the Duwamish/Green River watershed consisted almost entirely of forests, wetlands, and floodplain sloughs (Collins and Sheikh 2005). With the arrival of settlers, towns and agricultural operations were established throughout the river valley in the mid 1800s (Sato 1997; King County 2005k). Logging, sawmill, and mining operations were scattered throughout the Green River watershed, particularly in the upper and middle Green Rivers. Meat packing industries were located along the middle Green River, and manure, blood, and floor washings were often discharged directly into the river (Foster 1945). Sedimentation from logging and mining operations, as well as human sewage, dairy farm drainage, and household garbage were also sources of contamination to the Green/Duwamish River (Collins and Sheikh 2005; Foster 1945).

Mines and mining towns, including the town of Black Diamond, the Royal Reward Mine, and the St. Helens mining district, were established in the upper and middle Green River watersheds. Though most of these mines are either no longer in use or are mined only to a limited degree, mine drainage containing copper, arsenic, mercury, and iron may enter tributaries to the Green River and still has the potential to contaminate water and sediment in the Green River (Ecology 2002b).

The upper Green River watershed is protected under a watershed management plan as the drinking water supply for the City of Tacoma (Tacoma Water 2007); forestry remains the primary land use in this watershed (King County 2004a). In the middle Green River watershed, major land uses include forested land (approximately 27%), residential land (approximately 50%), and agricultural land (approximately 12%) (King County 2005k). In the lower Green River watershed, expansive urban development has been the most significant land use change from historical conditions, with approximately 50% of the land now residential, and 27% of the land developed for commercial and industrial use (King County 2005k). Agriculture, mixed land use, and park areas make up the remainder of the lower Green River watershed.

Although the upper and middle Green Rivers generally have good water quality, the lower Green River frequently exceeded Class A standards[146] for water quality (temperature, dissolved oxygen, and fecal coliform bacteria) (Washington Environmental Hearings Office 1996). Mercury had previously exceeded WQS for the Lower Green River but currently meets Category 2 standards; the Duwamish River currently meets Category 1 standards. Water bodies listed in Category 1 meet all of the tested standards, and water bodies listed in Category 2 are waters of concern.

Water bodies are listed under Category 5 if Ecology has data indicating that WQS have been violated, and there is no TMDL or pollution control plan. Water bodies in this category require TMDLs and are placed on the 303(d) list. The lower Green River and the LDW have been placed on Ecology's biannual Category 5 list under Section 303(d) of the federal CWA for fecal coliform bacteria, dissolved oxygen, and temperature (Ecology 2008e).

In 2003, King County conducted a WQA of the Green and Duwamish Rivers (Herrera 2005). The study spanned the area from the Howard Hanson Dam in the upper Green River watershed to the mouth of the Duwamish River at Elliott Bay, and included locations in the main river system and in selected tributaries. One sampling station was located in the upper Green River subwatershed, nine were located in the middle Green River subwatershed, six were located in the lower Green River subwatershed, one was located at the lower Green River/Duwamish River boundary, and one was located in the Duwamish River Estuary subwatershed. This assessment included analyses of nutrients, metals, and organic analytes (Herrera 2005).

Total and dissolved concentrations of arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc in the Green River watershed were compared with Washington State WQS for freshwater (Herrera 2005). Only mercury and zinc concentrations were above state standards at some stations. Of the 86 samples collected during base flow, the concentration of dissolved mercury was above the chronic standard in one sample each from the Duwamish River subwatershed and the lower Green River subwatershed. The concentration of dissolved mercury was less than the standard during storm flow at all sampling locations. Of the 58 samples analyzed for dissolved zinc, the concentration of zinc was above acute WQS in one storm-flow event sample from the Mill/Springbrook Creek (tributary entering at the lower Green River/Duwamish River boundary) (Herrera 2005). The concentration of dissolved zinc was less than the standard during base flow at all sampling locations.

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[146] The Washington 1997 WQS used a class-based system that separated each water body into a specific "class" depending on the beneficial use of that water body (EPA 2006a). The four classes in the system are Class AA, Class A, Class B, and Lake Class. Class A standards are protective of salmon and trout spawning and non-core rearing and migration (EPA 2006b). The Washington 1997 WQS were revised in 2003 and no longer consist of a class-based system.

During this study, aluminum, dissolved arsenic, cadmium, chromium, lead, mercury, dissolved nickel, and zinc concentrations did not vary significantly among upriver and downstream locations (Herrera 2005). Concentrations of total arsenic were statistically higher in downstream locations relative to upriver locations within the Green-Duwamish system. Dissolved copper and total nickel concentrations also increased significantly with distance downstream during storm flow conditions, but not during base flow conditions; the increased copper and nickel concentrations remained below applicable WQS (Herrera 2005). Although not reported as part of the 2003 WQA, a previous water quality investigation conducted in the Green/Duwamish River system between 2001 and 2002 had indicated a trend of increasing total aluminum, copper, lead, and zinc concentrations with distance downstream, and decreasing dissolved aluminum concentrations with distance downstream during storm flow conditions (Herrera 2005).

Nutrient and metal samples were collected using autosequential and autocomposite methods and occasionally as grab samples. Samples analyzed for organic compounds were collected only as grab samples, and therefore, less information is available compared with metals and nutrients (Herrera 2005). Organic compounds analyzed included PCB Aroclors, SVOCs (including PAHs and phthalates), VOCs, organophosphate pesticides, and chlorinated herbicides. VOCs and PCB Aroclors were never detected; however, the detection limits for PCB Aroclors were noted to be greater than the Washington State chronic WQS for freshwater for total PCBs (detection limits ranged from 0.04 to 0.06 µg/L, whereas the chronic freshwater quality standard is 0.014 µg/L).

Phenol was detected at low concentrations (less than 0.09 to 0.67 µg/L) at all sampling locations except for one location in the lower Green River where phenol was not detected. Phthalates and many PAHs were rarely detected; detection limits were not identified in source documentation. Multiple PAHs were detected in at least three samples from the Black River (a tributary that enters at the lower Green River/Duwamish River boundary). Organophosphate pesticides were not detected, though the detection limits were greater than the Washington State chronic WQS for freshwater for some pesticides (detection limits for organophosphate pesticides ranged from 0.29 to 0.52 µg/L). Of the chlorinated herbicides, atrazine was detected in one sample (0.98 µg/L in a storm flow sample) from Newaukum Creek (a tributary to the middle Green River), 2,4-D was detected in two samples from Springbrook Creek, a tributary to the lower Green River (0.1 µg/L in base flow and 0.37 µg/L in a storm flow sample) and in two samples from Newaukum Creek (0.08 µg/L and 0.19 µg/L in a storm flow sample), and dichlorprop was detected in one sample from Newaukum Creek (detection limits for chlorinated herbicides ranged from 0.015 to 0.046 µg/L) (Herrera 2005).

Water quality index scores for the Green River decreased from upriver to downstream, and concentrations of most analytes increased from upriver to downriver (Herrera 2005). The water quality index is a number intended to rank the water quality in certain areas. A higher water quality index score indicates better water quality (Herrera 2005). Water quality index scores were calculated for each of the Green/Duwamish River tributaries. Many of the tributaries were ranked as being of high concern for water quality, primarily because of high total nitrogen, total phosphorus, and fecal coliform bacteria concentrations, and because of low dissolved oxygen concentrations. Urban, residential, and agricultural land uses in the drainage basins of these creeks and streams were directly associated with the reduced water quality index scores (Herrera 2005). The water quality index at the Duwamish River Estuary subwatershed sampling station (Hamm Creek) was ranked as being of moderate concern.

King County performed water quality statistical and pollutant loading analyses on the data collected in 2002 and 2003. The loading analyses were used to derive correlations between constituent loadings and land use patterns in the Green River drainage basin (Herrera 2007). The loading analyses showed a correlation between agricultural land use and copper loading. Forested lands were not correlated with metals loading during base flow conditions but storm events could influence metals loading within those areas. The loading analyses also indicated that there was a correlation between the presence of roads and impervious surfaces and zinc and fecal coliform bacteria loading. Commercial areas were positively correlated with metals loading. Loading from low-density residential areas was highly variable, but negatively correlated with metals loading. Within the developed basins, metals loading was correlated with storm water. In general, the ratio of median storm-to-base flow concentrations of metals was greater than or equal to 1, but the ratios for dissolved metals were particularly high in Hamm Creek (4.5 for copper, 5.0 for mercury, and 4.0 for zinc) (Herrera 2007).

9.4.3.2 Creeks

Hamm Creek and Puget Creek are two significant open-channel systems that discharge to the LDW (Map 2-1). The creeks drain upland areas and have the potential to provide a pathway for chemicals to reach the LDW.

Puget Creek is a surface water body that drains primarily residential land, along with a small amount of commercial land (the Seattle Community College campus). The creek includes three convergent forks. The Puget Creek natural drainage area is approximately 294 ac; however, additional stormwater conveyances (e.g., ditches) increase the drainage area to approximately 430 ac (Rheaume 2007). There is generally year-round base flow to the north of Brandon Street, but the upper reaches are typically seasonal (Rheaume 2007). The creek primarily drains runoff from the surrounding residential neighborhood before it enters the SD system at W Marginal Way SW and flows into the LDW near T-105. Of the five risk driver chemicals discussed in Section 4 in more detail, only PCBs had sediment concentrations exceeding SMS criteria near the mouth of Puget Creek. The concentration of PCBs in surface sediment exceeded the SQS, but not the CSL, in at least one sample. No other exceedances of SMS criteria were detected.

Hamm Creek is a seasonal tributary to the LDW and has four forks: the north fork, the south fork, the middle fork, and the lost fork. Historically, the creek meandered through an intertidal marsh before discharging to the LDW. However, the creek was eventually connected to a piped system, which discharges to the LDW near S 96th St. The middle fork converges with the north fork near S 96th Street, west of Highway 99 and continues to discharge to the LDW via the S 96th Street storm drain outfall. In 2000, King County completed a habitat restoration project which rerouted the south fork to the waterway through a newly constructed stream channel (NOAA 2009a). The lost fork does not appear on maps in the RI because it is largely a piped system. In addition, Ecology determined that the lost fork of Hamm Creek is actually Durham Creek (Wood 2007). Durham Creek discharges to the waterway via the City's 7th Ave S storm drain outfall.

All four forks together drain approximately 1,408 ac (Kerwin and Nelson 2000) of residential, commercial, and industrial land. The north fork basin is more industrialized than the basin drained by the south fork (Herrera 2005). Hamm Creek flows through urban residential and portions of the West Duwamish Greenbelt before discharging to the LDW at approximately RM 4.2 (Kerwin and Nelson 2000).

The Hamm Creek drainage basin (including all of the forks mentioned above) is primarily residential (850 ac), with 316 ac of open space, including both the Rainier Golf and Country Club and Glen Acres Golf and Country Club. Open space is also occupied by public ROWs and some second-growth forest. The lower portions of the creek are heavily impacted commercial and industrial areas (242 ac). Water quality in all three forks of Hamm Creek has shown evidence of degradation (Kerwin and Nelson 2000). During base flow conditions, elevated pH values have been reported. These pH values are thought to be related to a cement kiln dust pile near S 96th Street and 10th Avenue (Kerwin and Nelson 2000). None of the chemicals analyzed in surface sediment near the mouth of Hamm Creek exceeded the SQS.

9.4.4 Direct discharge system information

This section presents an overview of direct discharges to the LDW, including permit requirements, management, and site-specific data. For the purposes of this report, direct discharge systems are defined as those that enter the LDW via a distinct pipe or outfall. In general, direct discharge systems include municipal or other publicly owned storm drain systems, municipal sanitary and combined sewer systems, and privately owned and managed storm drain systems. The main classifications of direct discharges include CSOs, EOFs, and public and private SDs.

In the area of the LDW, these discharge systems transport stormwater collected from industrial, commercial, and residential neighborhoods. These discharge systems also have the potential to transport treated wastewaters (from private industrial pre-treatment systems) and raw sewage during heavy rainfall events.

In May 2003, SPU conducted a low-tide outfall survey for the LDW study area to locate, inspect, and collect data on the horizontal and approximate vertical locations of apparent outfalls in the LDW. No attempt was made, however, to verify whether each and every pipe observed during the survey was in fact an active outfall. For example, some identified outfall locations have later been determined to be former water intake or pressure relief pipes and not outfalls. The survey also collected information on the size and composition of the outfall piping and estimated the rate of the flow (in gpm) exiting the pipe at the time of the survey, when feasible. The survey area included the east and west shorelines of the LDW from the south end of Harbor Island to approximately RM 5.0 (near the Norfolk CSO/SD) (Herrera 2004a).

For completion of the RI, the most up-to-date and readily available outfall-specific information was synthesized, including information on discharges identified in the Phase 1 RI, the 2003 outfall survey, and updated outfall information from Ecology, EPA, City of Seattle, City of Tukwila, Boeing, Port of Seattle, and King County records. Through this effort, 244 individual outfalls were identified within the LDW study area. To assist in this effort, Boeing also conducted an internal records review to further evaluate the current status and ownership of the outfalls on their properties along the LDW. Boeing completed an in-field inspection of the shoreline along Boeing Plant 2 to identify and confirm outfall locations noted in the 2003 survey. During this effort, Boeing identified additional outfall locations that had not been reported in the 2003 survey and noted other locations that were either abandoned or improperly classified as outfalls by Herrera (2004a).

Based on this information, the current status, use, and ownership of the outfalls discharging to the LDW were identified. These outfalls were identified and categorized using the classification nomenclature presented in Table 9-7. Maps 9-8a through 9-8f present the identified outfall locations, their classification, and ownership status. Detailed outfall-specific information is included in Appendix H.

Improvements and upgrades to stormwater networks and outfalls is a continuous process in the LDW basin. The outfall information discussed in this section and in Appendix H represents the best knowledge available from the resource documentation cited at the time that the outfall survey was conducted and reviewed. Some discrepancies exist between the outfall information from the LDW outfall survey and outfall information discussed in Ecology's SCAPs and data gaps reports. These discrepancies are generally the result of recent changes in outfall configuration or differences in the source documentation used to generate the information. Discrepancies between the LDW outfall survey and outfall information provided in SCAPs and data gaps reports for several of the SCAs are discussed in Appendix I.

Table 9-7. Outfall nomenclature and descriptions

Nomenclature	Description
CSO	Overflow point on a publicly owned and maintained combined sewer system. Combined sewers convey both wastewater (residential and industrial) and stormwater runoff. Overflows generally occur only during large storm events when the capacity of the combined sewer is exceeded and not all flow can be successfully conveyed to a treatment plant. Under these conditions, excess flow is discharged to a nearby receiving water body to prevent sewage backups.
EOF[a]	Overflow point on a combined or sanitary sewer, generally located at a pump station. Discharges are not storm related. Overflows occur as a result of mechanical failure, pipe obstruction, or power failure. Pump stations in LDW are equipped with backup generators.
CSO/SD or EOF/SD	Shared outfall that serves as both a combined sewer overflow or emergency overflow and storm drain system.
Public storm drain	Storm drains that are required to have coverage under an NPDES municipal stormwater permit (i.e., City of Seattle, King County, Port of Seattle, and WSDOT).
Private storm drain	Storm drains that collect storm and surface waters from areas in addition to those mentioned above. Discharges from these facilities may be covered under the general industrial NPDES permit, individual stormwater permits, or other site-specific requirements governing discharge to surface water bodies.
Stream, channel, or swale	Locations of open-air, convergent discharges into the waterway.
Abandoned outfall	Outfalls that are plugged or have existing documentation to show that the upland pipe has been disconnected.
Not an outfall	Location previously identified as an outfall in the 2003 survey; subsequent in-field investigation or identification of historical information led to a determination that the location was not an outfall (e.g., water intake or pressure release pipes).
Pipe of unresolved origin and/or use	Pipe identified as an outfall by Herrera (2004a) but with an undetermined origin and/or use.

[a] This designation is also used to identify an overflow feature from a reservoir, as appropriate. Reservoir overflows are not associated with discharges from the combined or sanitary sewer system.

CSO – combined sewer overflow

EOF – emergency overflow

LDW – Lower Duwamish Waterway

NPDES – National Pollutant Discharge Elimination System

SD – storm drain

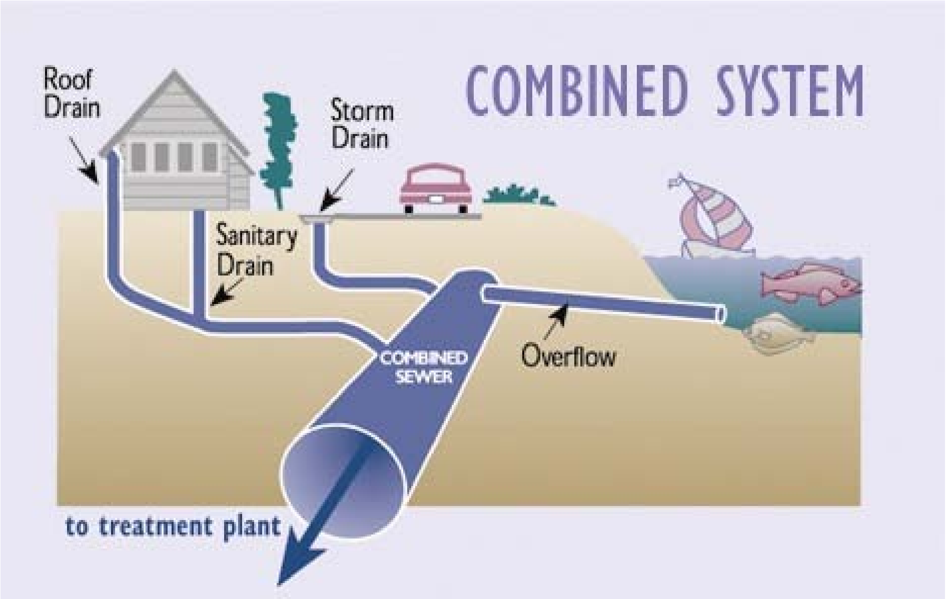
WSDOT – Washington State Department of Transportation

9.4.4.1 Combined sewer overflows and emergency overflows

Discharges from CSOs occur during rainfall events when the CSO system capacity is insufficient to transport the volume of both sanitary wastewater and stormwater flows. When this capacity is exceeded, excess flow is discharged to the LDW through an overflow structure (Figure 9-4). This flow can consist of a combination of untreated residential and industrial wastewater and stormwater runoff. CSOs are a recognized source of water pollution as a result of temporary increases in bacterial counts in the LDW, aesthetic degradation of the shoreline, and public health concerns in areas where human contact with the discharged water is possible.

An EOF is a discharge that can occur from either the combined or sanitary sewer systems that is not necessarily related to storm conditions and/or system capacity limitations. EOF discharges typically occur as a result of mechanical issues (e.g., pump station failures) or when transport lines are blocked. Pressure relief points are provided in the network to discharge flow under emergency conditions to prevent sewer backups. Pump stations are operated by both the City of Seattle and King County.

The total area of the combined CSO/SD basin for the LDW is 28,700 ac (Schmoyer 2009b). The City of Seattle and King County can potentially discharge combined untreated residential and industrial wastewater and stormwater runoff through CSOs to the LDW study area during these storm events. The City of Seattle owns and operates the local sanitary sewer collectors and trunk lines, while King County owns and operates the larger interceptor lines that transport flow from the local systems to the West Point WWTP.



Source: King County (2006a)

Figure 9-4. Schematic diagram of CSO system

Infrastructure improvements have greatly improved system storage capacity and reduced the number of discharges from the combined sewer systems (those that may include contributions of stormwater, sewage, and industrial waste streams). These combined systems are still in operation in some areas adjacent to the LDW, but their existence is very limited.

9.4.4.2 Storm drain systems

The LDW study area is also served by a combination of public and private storm drain systems. Many of the privately owned storm drain systems interconnect with public systems (including the combined sewer systems) before discharging to the LDW. In some cases, primarily for properties adjacent to the waterfront, private storm drain networks discharge directly to the LDW. The City of Seattle and King County have general knowledge of the location of these remaining systems; infrastructure upgrades are ongoing. The majority of the current volume of sanitary wastewater within the LDW basin is routed via public and private systems to the West Point WWTP.

For the purpose of this document, a public storm drain is defined as any system that is required to have coverage under a National Pollutant Discharge Elimination System (NPDES) municipal stormwater permit (i.e., systems owned and operated by the City of Seattle, Port of Seattle, King County, and the Washington State Department of Transportation [WSDOT]). All other drainage systems are classified as private.

During periods of rainfall, urban stormwater runoff may enter the LDW directly from roadways or properties adjacent to the waterway (i.e., direct stormwater runoff) or through these storm drain systems. Urban areas have the potential to accumulate particulate materials, dust, oil, asphalt, rust, rubber, metals, pesticides, detergents, and other chemicals resulting from urban activities. Chemicals present on pervious and impervious surfaces (e.g., roadways, parking lots, residential yards, industrial yard areas) can then be flushed into storm drains during wet weather and transported to the LDW in dissolved or particulate form. These drainage networks also provide a pathway for spills and leaks to reach the LDW.

An estimated 8,936 ac of land drain to the LDW via storm drains. Land use in the basin is summarized in Table 9-8. Table 9-9 presents LDW basin statistics based on geographic area and system ownership. Recent analyses by the City of Seattle and King County estimated approximately 4,000 mgy in discharges (including flow from Hamm and Puget Creeks) into the LDW through the storm drain and natural drainage networks (SPU 2008).

Table 9-8. Land use in the LDW drainage basin

LAND USE	AREA (ac)	PERCENTAGE OF TOTAL AREA
Commercial	529.0	5.9
Industrial	2,580.6	28.9
Multi-family residential	255.7	2.9
Schools	170.2	1.9
Parks, open space	1,306.7	14.6
Single family residential	1,773.8	19.8
Right-of-way	2,320.0	26.0
Total	**8,936**	**100.0**

Source: Schmoyer (2008d)
LDW – Lower Duwamish Waterway

Table 9-9. Relative size of sub-basin discharges to the LDW

SUB-BASIN	AREA (ac)	PERCENTAGE OF TOTAL AREA
City of Seattle municipal storm drain system	5,458	61
Public storm drains outside of City of Seattle[a]	1,865	21
Other storm drain systems and waterfront discharges within the Seattle city limits[b]	1,108	12
Private storm drain systems and waterfront discharges outside of Seattle city limits[c]	505	6
Totals	**8,936**	**100**

Source: Schmoyer (2008d)

[a] Includes storm drains owned and operated by the City of Tukwila and other public drainage networks in unincorporated King County.

[b] Storm drains located within Seattle that do not connect to the City of Seattle's municipal storm drain network (including systems owned by King County, the Port of Seattle, and other private owners along the LDW.

[c] Storm drains outside Seattle serving areas immediately adjacent to the waterfront, owned and operated by waterfront property owners.

LDW – Lower Duwamish Waterway

Annual stormwater discharge volumes are usually substantially higher than annual CSO discharge volumes because storm drains discharge whenever it rains, and CSOs only discharge during larger storm events. Annual stormwater discharges to the LDW have been estimated at approximately 4,000 mgy compared with less than 65 mgy historically from the county CSOs and less than 10 mgy from the city CSOs (Huber 2002; Schmoyer 2002; SPU 2007).

Private storm drain systems usually support smaller drainage basins (often less than 5 ac. In the LDW drainage basin, these smaller areas are usually immediately adjacent to the LDW and are located primarily on commercial and industrial use properties. An estimated 1,613- ac discharge to the LDW through these private systems associated with industrial facilities. In contrast, Seattle's municipal system drains about 5, 458 ac, and the publicly managed system outside of Seattle's basin (including the Hamm Creek drainage and the City of Tukwila's network) drains approximately 1,865 ac (SPU 2007).

9.4.4.3 Discharge permit programs

Wastewater and stormwater discharges from industrial facilities, public systems, and construction sites are each potential sources of contamination to the LDW. In general, discharge permit programs have been established by federal and state governments to help reduce the impact to water quality from these types of discharges. Both the state NPDES program and King County's Industrial Waste Program (IWP) regulate discharges to the LDW. Wastewater discharges are regulated by the NPDES and IWP programs; the NPDES program also regulates stormwater discharges from municipal separated storm drains, industrial sites, and construction sites. The following section discusses these various permit programs and how they apply to discharges to the LDW.

NPDES Programs

The NPDES program was established in 1972 under Section 402 of the CWA and requires a permit to discharge chemicals from any point source into waters of the United States. The NPDES program is administered by Ecology and regulates both wastewater and stormwater discharges to surface waters. Toxic chemicals, such as metals and man-made organics, as well as conventional parameters (5-day biological oxygen demand [BOD], TSS, pH, fecal coliform bacteria, and oil and grease) may be regulated under NPDES permits. Non-conventional parameters, including ammonia, phosphorus, chemical oxygen demand, and toxicity may also be regulated under the NPDES program (EPA 2002b).

There are two basic types of NPDES permits issued: individual and general. Facilities with similar operations, wastes, effluent limits, or monitoring requirements can be covered under general permits (EPA 2002b). Facilities with the same stormwater point source can also be covered under a general permit. NPDES general industrial stormwater permits, which are by far the most common, currently set limits for only a few parameters and chemicals (e.g., turbidity, pH, oil and grease, copper, and zinc). Under the CWA, all NPDES permits are issued for 5-year terms, and applications for renewal/reissuance are required not less than 180 days prior to the expiration of the current permit. NPDES-permitted discharges to the LDW include municipal stormwater (Phase 1 and Phase 2 municipal permits), general industrial stormwater, construction projects, boatyards, shipyards, CSOs, individual industrial wastewater (e.g., LaFarge, Seattle Iron and Metals), and sand and gravel operations.

King County and City of Seattle CSO discharges are permitted under individual NPDES permits. All King County CSOs located in the LDW are covered under the West Point WWTP permit (NPDES No. WA-002918-1) (Ecology 2005b). The City of Seattle's two CSOs are regulated under the NPDES permit No. WA-003168-2 (Ecology 2005c). City of Seattle and King County EOF events are reported to Ecology via Ecology's Environmental Report Tracking System (ERTS). Dry weather overflows (prohibited in NPDES permits), pump station failures, and blocked lines resulting from fats, oils, and grease; ragging;[147] or other reasons are routinely reported to Ecology's Water Quality Program.

Ecology has issued three types of general NPDES permits relevant to discharges to the LDW: one for sand and gravel operations, a second for boatyards, and a third for industrial stormwater operations at several facilities (Ecology 2007h). The general stormwater permit is also subdivided into three subcategories: municipal, industrial, and construction stormwater permits (Ecology 2007e).

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[147] Ragging is the build up of raggy components of raw sewage.

Sand and gravel general permits cover discharges of process water, stormwater, and mine-dewatering water (Ecology 2007p). Boatyard general permits are issued to commercial businesses that maintain, construct, or repair small vessels (Ecology 2007h).

There are two types of municipal stormwater permits defined under the CWA (40 CFR 122.26), based on the size of the municipality's population. Ecology's Phase 1 permits cover discharges from large and medium municipal separate stormwater sewer systems, except for municipal separate storm sewers owned or operated by WSDOT. The City of Seattle, the Port of Seattle, and King County are all covered under Phase 1 permits (Ecology 2007p). The first Phase 1 municipal stormwater permit did not require stormwater quality monitoring; however, monitoring of stormwater discharges is required under the new 2007 Phase 1 permit. Under the new permit, Phase 1 permittees are now required to monitor stormwater at three locations in their systems representing different land uses (i.e., low- and high-density residential, commercial, industrial for cities and commercial, low density residential, and high density residential for counties). Samples are required to be analyzed for a number of chemicals and parameters including many of the sediment COCs identified in the LDW, as well as for toxicity. The new Phase 1 permit also requires that permittees collect an in-line sediment sample at each monitoring location once each year during the monitoring program (Ecology 2007p).

Industrial general stormwater NPDES permits regulate stormwater discharges from industrial facilities. These facilities are required to monitor total copper, total lead, total zinc, oils and grease, turbidity, BOD, ammonia, nitrate-nitrite nitrogen, total phosphorus, and pH from at least one outfall location. All industrial facilities with industrial general stormwater permits are required to conduct quarterly monitoring of authorized discharges to surface water (Ecology 2007e).

Construction general stormwater NPDES permits are required for clearing, grading and excavation activities that affect 1 or more acres and that will discharge into surface water. Permits are also required if Ecology determines that an activity may potentially contaminate surface waters or if a water quality standard is expected to be violated (Ecology 2007a).

Facilities that do not qualify for a general permit must apply for an individual permit. Individual permits regulate the discharge of pollutants associated with a particular industrial or municipal facility. Individual permits have been issued to the City of Seattle and King County for CSO discharges (EPA 2002b).

A search was conducted to identify NPDES permitted discharges to the LDW. This search included a review of Ecology's Water Quality Permit Life Cycle System (Ecology 2007o). The system included facility addresses for each permittee but did not have information on outfall locations. A facility was assumed to potentially discharge to the LDW if the facility was within the LDW surface water drainage basin (according to the City's geographic information system [GIS] shape file). Six sand and gravel, sixty-seven industrial general, three construction general, three boatyard general, and five individual permitted facilities were identified within the LDW surface water drainage basin (see Table G-1).

NPDES Permits in the Vicinity of the LDW Study Area

It is possible that facilities discharging to the Duwamish River upstream of the LDW could also affect the water quality within the LDW. Table G-2 provides a summary of the facilities with an NPDES permit in the Black River, Duwamish River, lower Green River-East or lower Green River-West drainage basins (according to Washington State Department of Natural Resources GIS shape files). Sound Transit Light Rail also holds an individual NPDES permit for outfalls that discharge to the Duwamish River during construction (including areas upstream of and within the LDW). Only one of these outfall locations is within the LDW. A summary of the other Sound Transit Light Rail outfall locations is presented in Table G-3.

King County Industrial Waste Program

Under the CWA, local wastewater collection and treatment facilities were granted local pretreatment authority to regulate the types and amounts of chemicals discharged to their sanitary sewer systems. In April 1981, Ecology authorized Metro as the local pretreatment authority in Seattle and surrounding areas; this authority later shifted to King County through a series of changes in local government. King County established an IWP in 1969, through which permits could be issued to facilities that discharge to the sanitary sewer system. This program became an EPA-delegated program consistent with the requirements of the CWA in 1981.

The IWP enforces local discharge limits and regulates the discharge of industrial wastewater through the issuance and enforcement of discharge permits. The activities of the IWP serve to reduce industrial loading to the sewer system and to CSOs in the event of discharges to the LDW. Requirements may include pretreatment before discharge and/or best management practices (BMPs). King County enforces both federal standards and local limits, whichever are more stringent. Local limits were established to protect sewer facilities and treatment processes, public health and safety, and waters receiving the discharge to enable King County to comply with its own NPDES permits. Regulated materials include heavy metals; flammable materials; sulfides; cyanide; pH; fats, oils, and grease; and organic compounds. Since it began, the program has achieved significant improvements in the quality of wastewater received by King County's WWTPs and thus also in CSO discharges (King County 2006c).

The IWP issues industrial wastewater discharge permits and authorizations to companies that have industrial processes with the potential to adversely affect King County's WWTPs. Permits almost always require companies to monitor their own discharges. Waste discharge permits are issued to federally regulated industries or to facilities with discharge volumes greater than 25,000 gpd. Major discharge authorizations are issued for discharges between 5,000 gpd and 25,000 gpd. Minor discharge authorizations are issued for discharges between 1,000 and 5,000 gpd. Letters of authorization are issued for discharges less than 1,000 gpd. A list of industrial facilities that currently hold industrial waste discharge permits and authorizations governed under the IWP is provided in Table G-4 in Appendix G.

In support of this program, King County staff inspect facilities with discharge permits at least once per year and sample wastewater from all permitted companies at least twice per year. Follow-up inspections and wastewater sampling are conducted to verify that identified conditions causing violations are corrected and eliminated. King County bases the enforcement response to each violation on a number of factors, including the severity and duration of the violation and the discharge and enforcement history of the facility or business (King County 2006b). Facilities demonstrating compliance with the County's permit can be issued discharge authorizations, which reduce the sampling regularity. King County inspects companies with discharge authorizations at least once every 5 years, but does not regularly sample them, relying instead on self-monitoring at these companies.

Hazardous Waste Management Inspection Programs

The regional Hazardous Waste Management Program (HWMP) complements King County's IWP by educating local residents and small businesses on methods to reduce hazardous waste and prevent water pollution. The program is a cooperative effort among King County's Department of Natural Resources and Parks (the Solid Waste and Water and Land Resources Divisions), Public Health, SPU, and 38 cities throughout King and Snohomish Counties. The program implements the local HWMP adopted in 1990 by King County and all of the regional cities.

The regional HWMP targets industry groups and geographic areas to provide technical assistance. Program staff make site visits to small businesses throughout King County and all of its incorporated cities to observe operating practices. When problem materials, such as lead, mercury, and solvent-based paints, are found to be disposed of in the sanitary sewer, program staff advise the company on correct practices. When necessary, staff can refer the matter to the IWP for regulatory action. In 2000, more than 3,000 businesses were inspected. Follow-up inspections indicated that 75 to 80% of businesses made at least one positive change in hazardous waste management or environmental practices as a result of the initial visit (King County 2002c citing Galvin 2001). The LDW area is included in the general coverage of the program; additional efforts in the LDW targeted auto body and repair shops, machine shops, photo laboratories, and dry cleaners in the basin.

The regional HWMP will conduct follow-up activities in the Duwamish/Diagonal drainage basin to support the early action cleanup for the Duwamish/Diagonal EAA consistent with the LDW source control strategy. For example, a list of businesses previously inspected is being compared with a list of current businesses in the Diagonal CSO/SD drainage basin to identify any new inspections that need to be made. This information will be used to identify the need for new site visits and to coordinate with the inspection data from both the stormwater and industrial waste inspection programs.

Current efforts are focusing on visits to new (and previously visited) businesses within the Diagonal CSO/SD drainage basin or businesses in the drainage basins of other EAAs.

Local Construction Stormwater Management Programs

In addition to the NPDES construction stormwater permits described above, local municipalities also regulate the discharge of stormwater and water from excavated soils (i.e., groundwater that "dewaters" from soils moved about or removed from a construction site). Surface water management is an important aspect of most construction operations. Depending on the size and complexity of the effort, large volumes of stormwater runoff can collect at a construction site that can be very high in TSS and can potentially contain chemical contaminants. This surface water may have to be temporarily stored and potentially pretreated prior to discharge to the storm drain network or sanitary sewer.

In the City of Seattle, the Department of Planning and Development (DPD) regulates discharges from construction sites to the storm drain and combined sewer systems per the stormwater, grading, and drainage control code (Seattle Municipal Code 22.800), which requires developers to control soil erosion and runoff from all construction projects. Depending on the size of the construction site, developers are required to either implement standard erosion and control procedures or submit a temporary erosion and sediment control plan for DPD review. Projects that will conduct a significant amount of excavation dewatering are also required to submit a dewatering plan for review and approval. Construction dewatering involves the removal of groundwater and accumulated stormwater encountered during excavation activities. This water can be contaminated depending on the environmental conditions of the site. Depending on the location of the project, stormwater and dewatering water may be treated and discharged to the storm drain system or discharged directly to the combined sewer system. Discharges to the combined sewer must obtain a Side Sewer Permit from the City of Seattle and comply with King County's IWP requirements.

In unincorporated King County, the King County Department of Development and Environmental Services regulates construction discharges (King County 2006b). Most construction runoff is discharged to nearby storm drain systems or directly to adjacent surface waters. Although there are separate departments responsible for regulation within the city and county limits, the City of Seattle and King County work together to regulate discharges during construction because the systems are interconnected.

King County's IWP regulates construction discharges when those discharges will enter the sanitary or the combined sewer system. A temporary dewatering permit is required for excavations greater than 12 ft deep, projects that disturb more than 1 ac, projects that encounter surface and/or subsurface water during construction, projects in an environmentally critical area, and disposal of contaminated surface and/or subsurface water. Water from dewatering operations can only be discharged to the public sewer system if authorized by the County. All dewatering discharges must meet County permitting requirements regardless of the discharge volume and period of discharge (King County 2006b). Stormwater discharged into the sanitary or combined sewer systems must contain less than 7 mL/L of settleable solids to prevent sewer lines from becoming restricted or blocked. The discharge of materials such as ashes, sand, grass, and gravel to the sewer system is prohibited, and entities that discharge these or other solids to the public sewer system are liable for any damages caused (King County 2006b).

Large-volume dischargers of construction-related water to the sanitary or combined sewer systems (i.e., more than 25,000 gpd) must find other means of disposal during the "wet months" (November through April) when the sewer system can be at full capacity. Discharges of construction-related water to the sewer system are not prohibited during the "dry months" (May through October).

9.4.4.4 King County/City of Seattle CSO management and permit compliance

An important element of source control is the use of discharge permits issued by regulatory agencies. These permits provide clear and enforceable requirements for controlling the frequency, volume, and duration of CSO discharges, and the chemical concentrations in those discharges. As part of the City of Seattle and King County NPDES permits, Ecology requires "the greatest reasonable reduction of combined sewer overflows at the earliest possible date" (WAC 173-245-010). Ecology also requires that CSO planning documents specify the means for complying with the regulations.

Under their permits, both King County and the City of Seattle are required to monitor the frequency, volume, and duration of CSO discharges, and the magnitude of storm events at the time of CSO discharges. Monitoring information is then submitted to Ecology in monthly discharge monitoring reports and annual CSO reports. The annual reports provide information on the annual frequency and volume of CSO discharges, discuss accomplishments achieved in reducing CSO discharges, and explain additional CSO reduction efforts that will be made in the coming year (WAC 173-245-090(1)). King County's NPDES permit also requires sediment monitoring but only at the West Point WWTP.

In addition to discharge monitoring reports and annual CSO reporting requirements, both the City of Seattle and King County's NPDES permits require the maintenance of long-term control plans and submittal of amendments to CSO reduction plans at the time of NPDES permit renewal. Management practices and policies, called the Nine Minimum Controls, detailed in both the City of Seattle and King County permits, must also be implemented in order to reduce CSO frequency, volume, duration, and pollutant loads. The Nine Minimum Controls require:

1. Implementation of operation and maintenance programs for the combined sewer systems
2. Maximized use of the storage capacity of the wastewater collection system to reduce CSO discharges

3. Use of CSO controls (City of Seattle) and pretreatment programs (King County) to minimize nondomestic discharges to CSOs
4. Maximization of wastewater flow to wastewater treatment plants during wet-weather events
5. Prohibition of CSO discharges during dry weather, with reporting requirements in the event of a dry weather CSO discharge
6. Control of solid and floatable materials in combined sewer system wastewater
7. Implementation of pollution prevention plans
8. Public notification of CSO events
9. Monitoring of CSO outfalls for discharges

King County's CSO control plan is periodically updated to incorporate current technologies and BMPs, and to integrate CSO control with other Wastewater Treatment Division capital improvement programs in order to monitor and reduce CSO discharges. Strategies incorporated into the control plans include managing pollution at its source, maximizing the use of existing system capacity, monitoring and modeling flows in the system, and constructing new CSO control facilities.

King County routes combined sewer flows that would otherwise be discharged during CSO events to the regional plants for secondary treatment or CSO (primary) treatment, when possible, in order to provide the highest level of treatment. In some areas of the system where flows cannot be conveyed to the regional plants, the flows are either conveyed to satellite CSO treatment facilities for treatment prior to discharge or they overflow at various CSO outfalls in the collection system. Flow control is achieved with the use of an automated control system that manages flows through the conveyance system so that during storm events, the maximum amount of wastewater is contained in pipelines and storage facilities until it can be conveyed to a treatment plant. CSOs discharge untreated wastewater only when flows exceed the capacity of these systems (King County 2006b).

Construction of King County CSO control facilities in the region began in the late 1970s. So far, about $320 million has been spent to control CSO frequency, volume, duration, and quality, and another $383 million is planned to implement the CSO control projects throughout King County over the next 25 years. Early projects involved sewer separation, flow diversion, and storage tunnels. Current and future projects involve construction of conveyance improvements, storage tanks, and additional treatment facilities. These new treatment facilities will provide preliminary treatment to remove large objects, sand, gravel, and other inorganic material; primary treatment to remove suspended solids; secondary treatment to remove soluble material and fine suspended solids; and disinfection to kill pathogens (King County 2008a).

King County plans to reduce volumes at six CSOs discharging to the LDW (Hanford at Rainier [Hanford No. 1], E Michigan, S Brandon Street, 8th Avenue W, W Michigan, and

Terminal 115 [T-115]). The plans include conveyance expansion, increased storage capacity, and treatment (such as the storage and treatment tanks included in the Norfolk CSO/SD and the S Brandon St CSO systems) (King County 2006b).

Since 1988, when monitoring and measuring of CSO flows began, flow control efforts have reduced CSO volumes from an estimated 2.4 billion gal. per year to approximately 900 mgy system-wide. Examples of King County CSO management projects that have either decreased CSO volumes or discharge frequencies include: 1) replacement of the pipe that leads from the Brandon Street Regulator Station to the EBI, providing some reduction in the volume of overflows, and 2) installation of portable monitoring within the Duwamish and West Seattle pump stations to better characterize conditions at the pump stations and siphons to improve flow and system capacity management (King County 2006b).

The City of Seattle's permit also requires completion of a CSO supplemental characterization study and a sediment survey report. The supplemental characterization study is to be completed for a subset of the City's permitted CSOs, and will involve collecting whole-water samples of the discharges from the subset of CSOs and analyzing those samples for conventional and priority pollutants. Solids samples will not be collected as a part of this study (Taylor Associates 2007). The final report is due to Ecology in December of 2009. The sediment survey report will use available sediment data collected in the vicinity of the permitted CSO outfalls to assess sediment quality and determine whether additional sampling is needed.

The City of Seattle has also made progress in controlling the volume, frequency, and duration of CSOs. Since 1980 (the baseline for City CSO modeling), the City has reduced the annual volume of CSOs from approximately 306 million gal per year to about 214 million gal per year (estimated average for years 1998 through 2007) (Lee 2009). In its 2001 CSO reduction plan update, SPU identified the Diagonal Avenue S CSO/SD as one of the six priority outfalls in the city; no other uncontrolled CSOs have been identified in the LDW (City of Seattle 2002; CTE 2001). In 2005, SPU replaced an undersized pipe on the largest overflow point in this system (Diagonal Avenue S CSO/SD No.111 D) to allow more water to enter the King County sanitary sewer system. The Diagonal Avenue S CSO/SD system is currently being monitored to determine whether additional storage capacity is needed to adequately control overflows.

9.4.4.5 Compliance and inspection programs

Compliance and inspection programs are conducted by various agencies and NPDES permit holders to identify and manage potential sources of chemicals that are typically generated from business practices within the LDW drainage basin. Ecology NPDES inspections assess compliance with permit requirements, which may include BMPs. Inspections conducted under the joint SPU/KC program note NPDES issues but also consider stormwater, spill control/response, industrial waste, and hazardous waste issues. Local inspectors work with businesses to implement appropriate corrective actions necessary to comply with local stormwater and wastewater codes. In addition to these broader, programmatic compliance issues, SCWG members such as the City of Seattle, the City of Tukwila, the Port of Seattle, and King County use these programs to bring additional focus to source control concerns in the LDW. Most of their compliance and inspection programs pertain to discharges to the storm drain and /combined sewer systems. The measures implemented jointly or by each member are described in greater detail below.

Business Inspection Programs

In 2001, SPU began a city-wide business inspection program to enforce the source control requirements of the City of Seattle's stormwater, grading, and drainage control code (SMC 82.200). Business inspections for the LDW source control program, which began in 2003, are part of the City's overall source control program, but focus on a wider range of pollution-generating activities, including hazardous waste and industrial wastewater management. Prior to 2001, the City conducted periodic stormwater inspections for specific projects in Elliott Bay and the Duwamish River to control sources of chemicals entering city-owned storm drains. Business inspection locations are shown on Map 9-9.

One of the earliest inspection efforts in the LDW occurred in the mid 1990s under a Centennial Clean Water Fund grant from Ecology (She 1997). Approximately 100 businesses in the LDW were inspected. The study focused on outdoor activities to minimize the presence of chemicals used or generated by onsite activities that could come in contact with stormwater runoff. Inspectors worked with local businesses to improve their pollution prevention practices and developed information bulletins for businesses to promote BMPs, including disposal/storage activities, landscape maintenance, and housekeeping practices, and to increase local awareness of the importance of protecting water quality. A number of measurable actions by businesses were noted as a result of the inspections, including movement of barrels indoors, improvement of housekeeping practices, reducing or diverting contaminated water entry away from storm drains, and other similar actions.

In 2001, SPU inspected 200 businesses in the western portion of the Diagonal drainage basin (109 drive-by inspections and 91 complete onsite inspections). A total of 81 of the businesses inspected were not in compliance with City stormwater source control requirements. Most of the problems were related to inadequate maintenance of onsite storm drainage systems (33%) and inadequate spill response programs (47%). SPU inspectors worked with the business owners to improve their stormwater pollution prevention practices.

In 2003, SPU and King County initiated a joint inspection program to support LDW source control efforts. The joint program addressed various potential pollution-generating activities, including stormwater management, hazardous waste management, and industrial waste disposal. This joint program was designed to minimize the disruption to businesses by conducting a single inspection that covered all of the municipal environmental regulations rather than having representatives from each program conduct individual inspections. All inspections were conducted under the existing code authorities of each jurisdiction. Inspectors from King County and SPU conducted joint inspections to cross-train inspectors from the various King County/City of Seattle programs about each other's program requirements. Later, each inspector worked independently. Since 2005, SPU has assumed all inspection duties for the LDW program, with King County providing technical assistance, as needed. King County also continues to conduct routine inspections as part of its ongoing programs (e.g., industrial waste).

As of June 2008, a total of 1,175 businesses in the LDW study area have been inspected (882 full onsite inspections and 293 screening inspections). In addition, over 826 follow-up inspections have been conducted to confirm that property owners have implemented the necessary corrective actions (Schmoyer 2008b).

Corrective actions were required at 540 of the 882 sites (61%) where full inspections were conducted. Inadequate maintenance of onsite drainage facilities was the most common problem (57% of the corrective actions). Problems related to spill control, containment, and response were the most common problem (42% of the corrective actions). Approximately 38% of the corrective actions required were associated with stormwater practices (e.g., illicit discharges/connections, structures that require cleaning or repair, improper storage of materials, poor housekeeping practices), The other problems identified during the inspections involved improper storage, handling, or disposal of hazardous materials and products (19% of the corrective actions) (Schmoyer 2008c).

To date, SPU has inspected businesses in the Duwamish/Diagonal Way EAA, the Slip 4 EAA, the T-117 EAA, the Norfolk EAA, the Trotsky Inlet EAA, the Slip 1 SCA, the St. Gobain to Glacier NW SCA, the Slip 2 to Slip 3 SCA, the Slip 3 to Seattle Boiler Works SCA, the Seattle Boiler Works to Slip 4 SCA, the Glacier Bay SCA, and the Riverside Drive SCA. The business inspection program has focused on reaching businesses that discharge to the LDW via the publicly owned storm drain or the combined sewer systems.

In 2009, SPU will continue conducting follow-up inspections in the Diagonal/Duwamish, Norfolk, Glacier Bay, Trotsky Inlet areas, as needed, focusing on higher-priority businesses. In addition, inspections will also expand into other areas to support sediment remediation efforts and Ecology's development of SCAPs for the remaining SCAs. In 2009, the City of Seattle expects to begin inspecting areas on the west side of the LDW that have not yet been inspected (Schmoyer 2009a). The City of Seattle will continue conducting business inspections to support the LDW source control program and will refer identified source control issues to King County and Ecology, as appropriate.

City of Tukwila and King County

The City of Tukwila has jurisdiction over a portion of the drainage in the southeast part of the LDW study area. King County has jurisdiction over a very small strip of land adjacent to the southwest part of the LDW study area. These entities will be brought in as appropriate to conduct any necessary stormwater source control activities consistent with the source control strategy. In 2007, the City of Seattle signed a memorandum of agreement with the City of Tukwila to allow the City of Seattle to inspect five high-priority sites in the Norfolk drainage basin that are in Tukwila's jurisdiction.

Port of Seattle

The Port of Seattle is implementing a tenant audit and inspection program for its approximately 30 tenants, aimed at minimizing the potential for stormwater impacts. Tenant activities include container storage and handling, shipping, fishing vessel moorage, barge loading, and marina operations. The Port will use inspection opportunities to observe activities that can affect stormwater discharges and oversee the implementation and operation of appropriate BMPs. These inspections will also allow Port inspectors to observe any hazardous/dangerous waste-generating activities, inspect usage and storage of hazardous materials, and investigate any other environmental concerns, including air quality, noise, and impacts on neighbors and surrounding communities.

As part of their lease, the Port of Seattle requires tenants to apply for and obtain their own permits for stormwater management and air discharges and to write their own plans for industrial discharges to King County sewers and any spill prevention, control, and countermeasures. The Port of Seattle reviews all the permits and plans for adequacy and implementation to protect the Port from potential liabilities and to ensure that appropriate environmental/public health concerns have been addressed. The Port of Seattle also communicates with federal, state, and local agencies to make sure that any potential concerns or actions are addressed.

The Port of Seattle has a multi-phased tenant compliance program that includes an environmental review of all new and renewed tenant leases, a walk-through of all new tenant facilities upon occupancy and exit, and depending on their activity, periodic inspections (weekly, monthly or annual). Most Port tenants do not have industrial activities that require a formal intensive checklist and environmental audit by the Port. The Port has a cooperative compliance program through which they communicate potential concerns with tenants, including responsiveness and technical assistance. For each tenant inspection, the Port of Seattle completes a tenant inspection form. The information on the completed form is entered into the Port of Seattle's electronic database for compliance tracking. The information in this database is not published as an independent report and was not available for summation in this document.

Ecology's Urban Waters Initiative

The LDW is one of three areas in the state selected in 2007 by the legislature under the new Urban Waters Initiative. The initiative is designed to provide resources to state and local agencies to control sources of pollution to selected areas. The goal of the initiative is to develop effective lines of communication with business owners and managers to help in the battle against pollution in the state's waterways (Ecology 2008d).

As part of this initiative, Ecology and local agencies have built upon the original SPU/King County joint program and been able to assign more inspectors to work with businesses in areas that affect the LDW. Inspectors under this initiative include Ecology's Hazardous Waste and Water Quality Programs, King County's Wastewater Treatment Division's IWP, and SPU. These coordinated inspections evaluate commercial and industrial properties for compliance with necessary permits and established regulations. The inspections determine if source control measures should be implemented and whether site-specific source tracing samples should be collected (Ecology 2008d).

9.4.4.6 CSO and storm drain inputs and loading information

As mentioned in the previous sections, the City of Seattle and King County are required by their NPDES permit to conduct monitoring to support overall permit compliance requirements. In some instances, other parameters are also monitored to aid in CSO program management with the goal of reducing the potential for CSO events and chemical contributions to the waterway. The following section presents the available data for both CSO and storm drain discharges in the LDW basin.

CSO/EOF Inputs to the LDW

Several CSO/EOF outfalls are located within the LDW study area (Maps 9-8a through 9-8f). The City of Seattle owns and operates two CSOs (S Brighton St CSO and Diagonal Avenue S CSO) and three EOFs (EOF 117/pump station 44, formerly the Slip 4 EOF, EOF 156/pump station 78, formerly the Isaacson EOF, and EOF 178/pump station 17, which discharges to the LDW at the Norfolk outfall). The City's two CSOs also operate as storm drains, which regularly discharge stormwater during rainfall events. The City's Diagonal Avenue S CSO/SD outfall carries overflows from both County and City CSOs and receives overflows from nine separate overflow points, including the County's Hanford No. 1 CSO. Map 9-1 shows the basin boundaries for the area served by sanitary and combined sewers that have the potential to discharge to the LDW through a CSO, EOF, or CSO/SD outfall location.

Table 9-10 presents the 1999-2005 annual overflow volumes and flow frequencies available for King County's CSO/EOFs plus the contributions of the County's Hanford No. 1 CSO to the City's Diagonal Avenue S CSO/SD outfall (Map 9-8a). Table 9-10 also presents information on annual sediment load information for King County and City of Seattle CSOs. The City of Seattle's Diagonal Avenue S CSO is the only city CSO with significant discharges. The other city CSO (S Brighton St) discharges only during extreme storm events, generally once every 5 yrs (Tetra Tech 1988a). SPU records show that the S Brighton Street CSO has not overflowed since monitoring began in March 2000 (King County and SPU 2005). King County uses the period between 1981 and 1983 as the baseline for measuring progress in controlling CSOs; baseline volumes were determined using computer modeling (King County 2006a).

Table 9-10. CSO/EOF volumes, frequency of discharge, and annual sediment load information

CSO/EOF	Discharge No.	Map ID	Owner	Annual Average Volume (mgy)		Annual Average Frequency (events/yr)		Annual Sediment Load (MT/yr)[a]
				Baseline (1981 – 1983)	June 1999 – May 2005	Baseline (1981 – 1983)	June 1999 – May 2005	
8th Avenue S	040	2107	King County	8	0.00	6	0	nr
S Brandon Street	041	2223	King County	64	30.96	36	26.3	15.0
Diagonal Avenue S[b, c]	111	2155	City of Seattle	nr	8.64[d]	nr	20	3.4
Duwamish pump station West	034	na	King County	< 1	0.58	<1	1.0	0.1
Duwamish pump station East[e]	035	2153	King County	nr	0.67	<1	0.17	1.0
Hanford No. 1[f]	031	2155[e]	King County	nr	10.36	nr	5.50	4.9
E Marginal Way S pump station[e]	043	na	King County	< 1	0.00	< 1	0	nr
E Michigan	039	2502	King County	190	19.01	34	8.14	9.0
W Michigan	042	2506	King County	2	0.98	5	3.57	0.5
Norfolk[c]	044	2095	King County	39	0.28	20	1.14	0.1
Terminal 115	038	na	King County	2	3.17	4	2.00	1.5

Source: King County (2006b), Nairn (2007), SPU (2007), and Schmoyer (2007a), (QEA 2008)

[a] Information provided in the STM report (QEA 2008).

[b] Volume/frequency include only contributions from the City-owned system.

[c] Combined CSO/SD.

[d] Annual average calculation does not include 59 million gal. discharge event that occurred October 20 and 21, 2003. Annual average volume calculation with that single event included is approximately 15.84 mgy.

[e] Emergency overflow.

[f] Hanford No. 1 discharges through the City's Diagonal Avenue S SD outfall, which also discharges the City's Diagonal Avenue S CSO.

CSO – combined sewer overflow
EOF – emergency overflow
MT/yr – metric tons per year
mgy – million gallons per year
na – not available
nr – not reported

As shown in Table 9-10, the E Michigan CSO, S Brandon Street CSO, and Hanford No. 1 (discharging via the City's Diagonal Avenue S CSO/SD) outfalls had the highest annual average overflow volumes between 1999 and 2005. In 2004, the Duwamish West pump station was reactivated under King County's NPDES permit because of concerns that it could overflow; monitoring of this pump station is now in place. The E Marginal Way S Pump Station and Duwamish East CSO/EOF are considered to be controlled (i.e., they meet Ecology's standard of one overflow event per year based on a 5-yr average) (King County 2006b).

King County's E Marginal Way S outfall is a pump station EOF; discharge at this station occurs only under emergency conditions (i.e., power or mechanical failure). King County has no record of an overflow event ever occurring at this location. The Duwamish East CSO/EOF also functions as an emergency bypass for a pump station; there has not been an emergency overflow since 1989 (King County 2006b). However, this location has the potential to overflow if the input volume from the siphon carrying combined sewage from the western bank of the LDW and the West Seattle area is too high (King County 2006b).

Whole-Water Sampling of CSOs

As part of two different studies, whole-water samples were collected from King County CSO discharges to the Duwamish River (King County 1999e). Samples were collected from the Norfolk CSO once in 1995 as part of the pilot study for the King County WQA and four times in 1997 as part of the Henderson/MLK CSO control project. The S Brandon Street CSO was sampled in 1996-1997 during 15 different CSO events (with more than one sample collected per event) for the King County WQA. Both of these CSOs discharge into the LDW study area; Tables 9-11 and 9-12 summarize the data collected from the Norfolk and S Brandon St CSOs (Williston 2008).

In the whole-water samples collected from the Norfolk and S Brandon Street CSOs between 1995 and 1997, six metals (i.e., arsenic, chromium, copper, lead, nickel, and zinc) were detected in 100% of the samples. Other constituents regularly detected included other metals, phthalates, and several PAHs. Other organic compounds, including PCBs and DDTs, were not detected in any of these CSO whole-water samples (Williston 2008). Surface sediment data for areas near these outfalls are discussed in relation to the SQS in Section 4.2.1 (Maps 4-14a through 4-14f). Source tracing samples collected within the Norfolk CSO/SD drainage basin (i.e., ROW catch basin, onsite catch basin, in-line sediment grab) are presented in Section 9.4.4.7.

Table 9-11. Chemical concentrations in Norfolk CSO discharge

PARAMETER[a]	DETECTION FREQUENCY	NUMBER OF SAMPLES	RLS FOR NON-DETECTED CONCENTRATIONS (µg/L) MINIMUM	RLS FOR NON-DETECTED CONCENTRATIONS (µg/L) MAXIMUM	NUMBER OF DETECTIONS	DETECTED CONCENTRATIONS (µg/L) MINIMUM	DETECTED CONCENTRATIONS (µg/L) MAXIMUM	DETECTED CONCENTRATIONS (µg/L) MEAN	DETECTED CONCENTRATIONS (µg/L) MEDIAN	DETECTED CONCENTRATIONS (µg/L) STANDARD DEVIATION
Antimony (total)	100%	3	na	na	3	0.56	1.2	0.87	0.85	0.32
Arsenic (total)	100%	3	na	na	3	4.1	4.9	4.6	4.7	0.43
Barium (total)	100%	3	na	na	3	21	57	44	49	20
Cadmium (total)	67%	3	0.20	0.20	2	0.28	0.31	0.30	0.30	0.021
Chromium (total)	100%	3	na	na	3	1.4	11.3	7.8	9.2	5.5
Cobalt (total)	100%	3	na	na	3	0.5	3	2.1	2.4	1.3
Copper (total)	100%	3	na	na	3	13	30	21	21	8.8
Lead (total)	100%	3	na	na	3	4.2	29	19	21	13
Mercury (total)	50%	4	0.20	0.20	2	0.21	0.25	0.2	0.23	0.03
molybdenum (total)	67%	3	0.50	0.50	2	0.59	0.98	0.79	0.79	0.28
Nickel (total)	100%	3	na	na	3	3.5	12	8.4	9.3	4.3
Silver (total)	33%	3	0.30	0.30	1	0.88	0.88	0.88	0.88	na
Vanadium (total)	100%	2	na	na	2	2.3	14	7.9	7.9	8.0
Zinc (total)	100%	3	na	na	3	38	95	71	75	29
Chrysene	20%	5	0.14	0.28	1	0.2	0.2	0.2	0.2	Na
Fluoranthene	60%	5	0.14	0.28	3	0.2	0.27	0.2	0.22	0.038
Phenanthrene	60%	5	0.14	0.28	3	0.17	0.33	0.2	0.20	0.09
Pyrene	60%	5	0.14	0.28	3	0.18	0.27	0.2	0.23	0.04
BBP	80%	5	0.14	0.14	4	0.26	1.23	0.58	0.51	0.44
BEHP	20%	5	0.28	1.71	1	4.7	4.7	4.7	4.7	Na
Diethyl phthalate	100%	5	na	na	5	0.28	2.2	1.2	1.2	0.73
1,4-Dichlorobenzene	100%	5	na	na	5	0.38	0.83	0.51	0.44	0.18
4-Methylphenol	60%	5	0.24	0.27	3	0.25	2.9	1.4	1.2	1.4
Benzoic acid	60%	5	0.94	0.95	3	1.2	2.5	2.1	2.3	0.75
Benzyl alcohol	40%	5	0.24	0.47	2	0.33	0.38	0.36	0.36	0.04

Table 9-11, cont. Chemical concentrations in Norfolk CSO discharge

PARAMETER[a]	DETECTION FREQUENCY	NUMBER OF SAMPLES	RLs FOR NON-DETECTED CONCENTRATIONS (µg/L)		NUMBER OF DETECTIONS	DETECTED CONCENTRATIONS (µg/L)				
			MINIMUM	MAXIMUM		MINIMUM	MAXIMUM	MEAN	MEDIAN	STANDARD DEVIATION
Coprostanol	80%	5	1.90	1.90	4	15	67	32	30	25
TOC	100%	4	na	na	4	8.8	16	13	13	3.2
TSS	100%	5	na	na	5	51	167	110	105	55

Source: (Williston 2008)

Note: Tentatively identified compounds and select conventional parameters were excluded from this table.

[a] All other compounds analyzed (including PCBs, 15 PAHs, 41 other SVOCs, 3 phthalates, and 10 metals) were not detected in these CSO samples.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
CSO – combined sewer overflow
na – not applicable
PCB – polychlorinated biphenyl
RL – reporting limit
TOC – total organic carbon
TSS – total suspended solids

Table 9-12. Chemical concentrations in S Brandon Street CSO discharge

PARAMETER[a]	UNIT	DETECTION FREQUENCY	NUMBER OF SAMPLES[b]	RLs FOR NON-DETECTED CONCENTRATIONS		NUMBER OF DETECTIONS	DETECTED CONCENTRATIONS (µg/L)				
				MINIMUM	MAXIMUM		MINIMUM	MAXIMUM	MEAN	MEDIAN	STANDARD DEVIATION
Antimony (total)	mg/L	100%	41	na	na	41	0.0012	2.3	0.0585	0.00205	0.36
Arsenic (total)	mg/L	100%	41	na	na	41	0.0014	3.5	0.089423	0.00284	0.55
Barium (total)	mg/L	100%	41	na	na	41	0.023	38	0.962273	0.0397	5.85
Cadmium (total)	mg/L	100%	41	na	na	41	0.00034	1	0.016492	0.00084	0.10
Chromium (total)	mg/L	100%	41	na	na	41	0.0045	7.9	0.207454	0.0104	1.23
Cobalt (total)	mg/L	100%	40	na	na	40	0.00095	2.3	0.060111	0.00227	0.36
Copper (total)	mg/L	100%	41	na	na	41	0.022	46	1.167466	0.0376	7.13
Iron (total)	mg/L	100%	24	na	na	24	2.66	5590	239.4925	5.435	1139.66
Lead (total)	mg/L	100%	41	na	na	41	0.019	48.8	1.244124	0.0407	7.61
Magnesium (total)	mg/L	100%	1	na	na	1	1.08	1.08	1.08	1.08	na
Manganese (total)	mg/L	100%	1	na	na	1	0.051	0.051	0.0513	0.0513	na
Mercury (total)	mg/L	15%	41	0.00	0.20	6	0.0002	0.00108	0.00046	0.00037	0.00
Mercury (total) (by CVAF)[c]	mg/L	100%	5	na	na	5	1.2E-07	8.56E-05	4.32E-05	3.15E-05	0.00
Molybdenum (total)	mg/L	100%	41	na	na	41	0.0032	4.38	0.115128	0.00672	0.68
Nickel (total)	mg/L	100%	41	na	na	41	0.0069	10.4	0.29568	0.0126	1.63
Selenium (total)	mg/L	29%	41	0.00	0.00	12	0.001	1.3	0.109508	0.0013	0.37
Silver (total)	mg/L	66%	41	0.00	0.00	27	0.00039	0.34	0.013655	0.00093	0.07
Sodium (total)	mg/L	100%	1	na	na	1	4.7	4.7	4.7	4.7	na
Tin (total)	mg/L	100%	4	na	na	4	0.0026	0.0035	0.00322	0.00339	0.00
Vanadium (total)	mg/L	100%	35	na	na	35	0.0031	6.74	0.199622	0.00684	1.14
Zinc (total)	mg/L	100%	41	na	na	41	0.12	156	4.023098	0.173	24.33
Butyltin (total)	µg/L	100%	1	na	na	1	4	4	4	4	na
Mono-n-butyltin (as monobutyltin ion)	µg/L	100%	1	na	na	1	4	4	4	4	na
2-Methylnaphthalene	µg/L	19%	32	0.38	0.80	6	0.39	1.7	0.76	0.64	0.49

Table 9-12, cont. Chemical concentrations in S Brandon Street CSO discharge

				RLs for Non-Detected Concentrations			Detected Concentrations (µg/L)				
Parameter[a]	Unit	Detection Frequency	Number of Samples[b]	Minimum	Maximum	Number of Detections	Minimum	Maximum	Mean	Median	Standard Deviation
Acenaphthene	µg/L	6%	32	0.09	0.20	2	0.12	0.19	0.155	0.155	0.05
Benzo(a)anthracene	µg/L	13%	32	0.14	0.30	4	0.15	0.3	0.18575	0.165	0.05
Chrysene	µg/L	75%	32	0.14	0.14	24	0.15	0.54	0.251083	0.234	0.08
Fluoranthene	µg/L	97%	32	0.14	0.14	31	0.15	0.72	0.308871	0.32	0.13
Fluorene	µg/L	16%	32	0.14	0.30	5	0.14	0.40	0.233	0.236	0.11
Phenanthrene	µg/L	81%	32	0.14	0.14	26	0.19	0.762	0.346692	0.284	0.14
Pyrene	µg/L	97%	32	0.14	0.14	31	0.15	0.812	0.321742	0.321	0.12
BBP	µg/L	34%	32	0.14	2.30	11	0.65	24	3.405545	1.14	6.82
BEHP	µg/L	78%	32	2.53	3.95	25	3.52	8.4	5.4328	5.21	1.10
Diethyl phthalate	µg/L	84%	32	0.24	0.24	27	0.26	1.6	0.654667	0.51	0.39
Dimethyl phthalate	µg/L	75%	32	0.09	0.20	24	0.096	0.31	0.158833	0.13	0.07
Di-n-butyl phthalate	µg/L	63%	32	0.24	0.60	20	0.32	18	1.9982	0.7215	3.91
Di-n-octyl phthalate	µg/L	100%	32	na	na	32	0.68	3.3	1.460438	1.2	0.63
1,4-Dichlorobenzene	µg/L	97%	32	0.15	0.15	31	0.15	2.69	0.516355	0.45	0.48
2,4-Dimethylphenol	µg/L	6%	32	0.24	0.50	2	0.37	0.42	0.395	0.395	0.04
2-Methylphenol	µg/L	38%	32	0.24	0.50	12	0.25	22	2.223333	0.37	6.17
4-Chloro-3-methylphenol	µg/L	22%	32	0.47	1.00	7	0.53	1.1	0.76	0.65	0.21
4-Methylphenol	µg/L	56%	32	0.24	0.50	18	0.25	26	2.984778	0.976	5.87
4-Nitrophenol	µg/L	59%	32	0.47	0.50	19	0.48	2.5	1.287368	0.96	0.73
Benzoic acid	µg/L	91%	32	0.94	0.94	29	0.96	23	5.328621	2.63	5.99
Benzyl alcohol	µg/L	47%	32	0.24	0.71	15	0.31	3.2	0.873467	0.504	0.85
Caffeine	µg/L	100%	32	na	na	32	1.23	29	6.700938	3.735	6.23
Coprostanol	µg/L	100%	32	na	na	32	8.5	96	31.60406	22.55	22.66
Isophorone	µg/L	13%	32	0.24	0.50	4	0.27	0.46	0.3675	0.37	0.08
Pentachlorophenol	µg/L	81%	32	0.24	0.50	26	0.25	0.597	0.370192	0.35	0.09
Phenol	µg/L	25%	32	0.94	2.00	8	1.2	16.1	3.92625	2.28	5.03

Table 9-12, cont. Chemical concentrations in S Brandon Street CSO discharge

PARAMETER[a]	UNIT	DETECTION FREQUENCY	NUMBER OF SAMPLES[b]	RLS FOR NON-DETECTED CONCENTRATIONS		NUMBER OF DETECTIONS	DETECTED CONCENTRATIONS (µg/L)				
				MINIMUM	MAXIMUM		MINIMUM	MAXIMUM	MEAN	MEDIAN	STANDARD DEVIATION
TOC	µg/L	100%	36	na	na	36	5.0	33	13	11	6.5
TSS	µg/L	100%	23	na	na	23	44	771	127	75	156
Ammonia nitrogen	mg/L	100%	35	na	na	35	0.053	3.1	0.605791	0.371	0.67
Nitrite + nitrate nitrogen	mg/L	94%	16	0.05	0.05	15	0.085	0.331	0.164267	0.134	0.08

Source: Williston (2008)

Note: Tentatively identified compounds and select conventional parameters were excluded from this table.

[a] Detection frequency was less than 5% for naphthalene; all other compounds analyzed (including PCBs,10 PAHs, 34 other SVOCs, 19 pesticides, 14 metals, and 2 organometals) were not detected in these CSO samples.

[b] Fifteen events were sampled, with most having multiple samples per event. Samples were first averaged per event (by date) and then summary statistics were calculated. For purposes of this summary table, all samples were used in the mean and median calculations and not combined per event.

[c] Five samples were analyzed for mercury using ultra trace level sampling techniques and analyzed by cold vapor atomic fluorescence.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
CSO – combined sewer overflow
CVAF – cold vapor atomic fluorescence
na – not applicable
PCB – polychlorinated biphenyl
RL – reporting limit
TOC – total organic carbon
TSS – total suspended solids

Lateral Loads Analysis for the LDW

In 2007, SPU estimated the lateral loads to the LDW for use in the STM that has been developed for the FS. The STM tracks the movement of sediment through the LDW from three primary sources: lateral loads, Duwamish/Green River contributions at the upstream boundary of the LDW, and re-suspension of in-waterway sediment. For the purposes of the STM, lateral loads are defined as the sediment contributions from small streams, storm drains, and CSOs along the LDW. The information used in the STM provides a general, large-scale perspective on lateral solids loading and transport in the LDW. These preliminary lateral load estimates may be refined in the FS.

The LDW drainage basin was divided into sub-basins for the lateral load analysis. The sub-basins were delineated based on City of Seattle GIS information as well as site-specific drainage plans (SPU 2007). In 2008, SPU updated the lateral loads analysis to incorporate source tracing data through December 2007 and updated sub-basin boundaries (see Map 9-10). Table 9-13 summarizes the updated TSS concentrations and stormwater discharge volumes estimated as part of the lateral load analysis (SPU 2008).

In order to estimate the volume of stormwater discharged to the LDW, the Hydrologic Simulation Program-Fortran was used to estimate runoff volumes for various combinations of land use, soil type, and slope within the drainage basin (SPU 2008). Data from CSO flow monitoring programs (1999 to 2005) were used to estimate discharge volume for CSOs. Flow estimates were calculated for a typical wet year (2002), dry year (1993) and an average year (1986) (SPU 2008). Rainfall data used in the model were collected at SPU's gauge station at E Marginal Way S and 13th Avenue S.

TSS concentrations in stormwater were estimated based on TSS data from 24 locations in Seattle, Bellevue, Tacoma, Issaquah, and Everett. Average TSS concentration calculations accounted for different types of land use. The TSS concentration was calculated based on the percentage of land use within each sample area (Table 9-8). Land-use weighted averages were multiplied by the calculated runoff volume for each land use category. TSS concentrations in CSO discharges were estimated using King County data from samples collected from 1995 to 1997 from five major CSO outfalls (Brandon, Chelan, Hanford, Connecticut, and King) (SPU 2008).

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Table 9-13. LDW storm drain estimated lateral load information

OUTFALL No.[a]	NAME	OUTFALL ID	DRAINAGE BASIN AREA (ac)	RUNOFF VOLUME			TSS LOAD		
				LOW WATER YEAR (mgy)	MED WATER YEAR (mgy)	HIGH WATER YEAR (mgy)	LOW WATER YEAR (MT/yr)	MED WATER YEAR (MT/yr)	HIGH WATER YEAR (MT/yr)
1	Diagonal Avenue S CSO/SD storm drain basin	2155	2,620	797	1,045	1,372	241	316	414
2	Norfolk CSO/SD storm drain basin	2095	826	317	416	545	91	124	162
3	Slip 4 SD drainage basin	2048	495	242	314	401	74	97	124
4	7th Avenue S SD drainage basin	2112	232	83	108	138	24	33	42
5	Seattle municipal storm drain (SW Idaho Street)	2147	655	158	211	285	42	62	83
6	Seattle municipal storm drain (SW Kenny/Glacier Bay)	2127	164	39	53	71	11	15	21
7	Seattle municipal storm drain (SW Highland Park Drive)	2125	491	118	159	214	42	62	83
8	Hamm Creek	2205	721	245	325	438	46	86	116
9	King County Airport storm drain No. 2 at former Slip 5	2062	236	119	154	196	35	48	61
10	King County Airport storm drain No. 1 at slip 6	2080	64	32	42	53	9	13	17
11	Remaining Seattle municipal storm drains (east side of LDW)	no number[b]	226	109	142	180	34	44	57
12	S 96th St SD	2100A	996	348	457	604	87	128	169
13	1st Avenue S channel	no number[b]	328	79	106	143	21	31	41
WF-1	Waterfront Area 1	na	46	23	30	38	7	9	12
WF-2	Waterfront Area 2	na	71	36	46	59	11	14	18
WF-3	Waterfront Area 3	na	43	22	28	35	6	9	11
WF-4	Waterfront Area 4	na	210	107	138	176	33	43	55
WF-5	Waterfront Area 5	na	101	50	64	82	14	20	26

OUTFALL No.[a]	NAME	OUTFALL ID	DRAINAGE BASIN AREA (ac)	RUNOFF VOLUME			TSS LOAD		
				LOW WATER YEAR (mgy)	MED WATER YEAR (mgy)	HIGH WATER YEAR (mgy)	LOW WATER YEAR (MT/yr)	MED WATER YEAR (MT/yr)	HIGH WATER YEAR (MT/yr)
WF-6	Waterfront Area 6	na	94	27	38	52	6	11	15
WF-7	Waterfront Area 7	na	37	17	23	29	5	7	9
WF-8	Waterfront Area 8	na	56	29	37	47	9	12	15
WF-9	Waterfront Area 9	na	47	24	30	39	7	9	12
WF-10	Waterfront Area 10	na	58	27	35	45	8	11	14
WF-11	Waterfront Area 11	na	120	49	65	86	13	18	24
Storm drain totals			8,936	3,096	4,065	5,328	878	1,224	1,600

[a] Waterfront areas represent sections of the LDW shoreline that were previously not identified as a component of the first 13 listed sub-basins. Waterfront areas were delineated by the City of Seattle. Although each waterfront area includes multiple outfalls, the loading analysis combined the contribution from each area as a single discharge for loading estimating purposes (SPU 2008).

[b] Outfall was not found during the 2003 outfall survey; no corresponding number is available (Herrera 2004a).

CSO – combined sewer overflow
LDW – Lower Duwamish Waterway
mgy – million gallons per year
MT/yr – metric tons per year
na – not applicable
SD – storm drain
TSS – total suspended solids

9.4.4.7 Storm drain source tracing information and control programs

Various source control efforts to identify and characterize potential sources of contamination have been implemented to minimize the potential for sediment recontamination in the LDW. As part of these efforts, source identification is being conducted in the LDW drainage basin. This source identification effort includes the collection of the following types of samples:

- Sediment from catch basins located on private property and within the public ROW
- Sediment from in-line sediment traps installed in the storm drain system
- In-line grab samples of sediment accumulated within storm drain lines collected from maintenance holes on the storm drain trunk lines
- Catch basin samples are grab samples of material that has accumulated in the catch basin sump. Catch basins are part of the stormwater collection system and collect runoff from a fairly small catchment area (< 1 ac). These structures are equipped with a small sump to capture sediment and other large debris before it can enter the stormwater conveyance system. The sediment that accumulates in catch basins provides a qualitative measure of the solid-phase stormwater load coming from a drainage system and may not reflect actual total loading to the waterway (whole water plus solids). Catch basin samples are collected either from a specific site or property (onsite) or from the streets (ROWs).

In-line sediment traps consist of a sample bottle mounted inside the collection system pipe. Traps are typically installed for a period of about 6 months to passively collect suspended particulates that pass by the sampling station. Sampling stations are selected to isolate specific drainage sub-basins or entire drainage basins (generally greater than 50 ac). SPU has installed in-line sediment traps in the storm drain systems that discharge to the Duwamish/Diagonal (6 traps) and Slip 4 (10 traps) EAAs (Map 9-2). In 2007, SPU also installed five in-line sediment traps in the Norfolk drainage system. However, the results from these traps were not available at the time this report was prepared.

In-line grab samples of sediment are samples of material that has accumulated within storm drain lines in quiescent areas where there is sufficient sediment present for chemical analysis. Like sediment traps, in-line grab samples also represent contributions on a basin-wide or sub-basin scale.

From January 2002 through December 2007, source tracing sediment samples have been collected from 171 onsite catch basins and 74 ROW catch basins in the LDW drainage basin. In addition, 130 in-line sediment grab samples and 102 in-line sediment trap samples have been collected (Schmoyer 2008a). Samples from onsite catch basins have primarily been collected from sites where business inspections were conducted and at sites where sufficient sediment was available for chemical analysis (see Map 9-11 for source tracing sampling locations through December 2007). Most of the catch basin samples have been collected in the Duwamish/Diagonal and Slip 4 drainage basins, but samples have also been collected from the Norfolk area, and from other sub-basins outside the Duwamish/Diagonal and Slip 4 drainages.

Samples from three catch basins, six manholes, one street dirt area at the western edge of the 16th Avenue S ROW, and one settling tank located upstream of an oil-water separator at the Basin Oil facility were also analyzed for dioxins (Integral 2008). These samples, which were analyzed for dioxin and furan congeners, were collected in 2004 and 2005 to support ongoing source tracing efforts in the LDW. Sample locations were selected to target areas with relatively high total PCB concentrations in source tracing samples previously collected (Integral 2008). TEQs were calculated using the mammalian TEFs as provided by van den Berg et al. (2006). Concentrations of individual congeners that were not detected were estimated at one-half the RL.

Dioxins and furans were detected in all 11 samples. TEQs in the three catch basin samples ranged from 11.3 ng/kg dw to 25.6 ng/kg dw, and TEQs in the manhole samples ranged from 6.2 ng/kg dw to 26.3 ng/kg dw (Integral 2008). The sample collected from the settling tank near Basin Oil had a TEQ of 15.2 ng/kg dw and the street dirt sample had a TEQ of 90.5 ng/kg dw. These data are not evaluated further in this section.

No regulatory standards exist for in-line storm drain sediment. Chemical concentrations in storm drain sediment were compared in this section to the SMS criteria and MTCA Method A cleanup standards for informational purposes only (MTCA methods were used for comparison of TPH values only). This comparison provides an estimate of storm drain sediment quality. TOC information was not available for all samples with detected concentrations. Detection frequencies were based on the total number of samples collected in each sub-basin per media type. For organic compounds with OC-normalized SMS criteria, percentages of samples with concentrations above the SQS and the CSL were based on the number of detected samples for which TOC information was available.

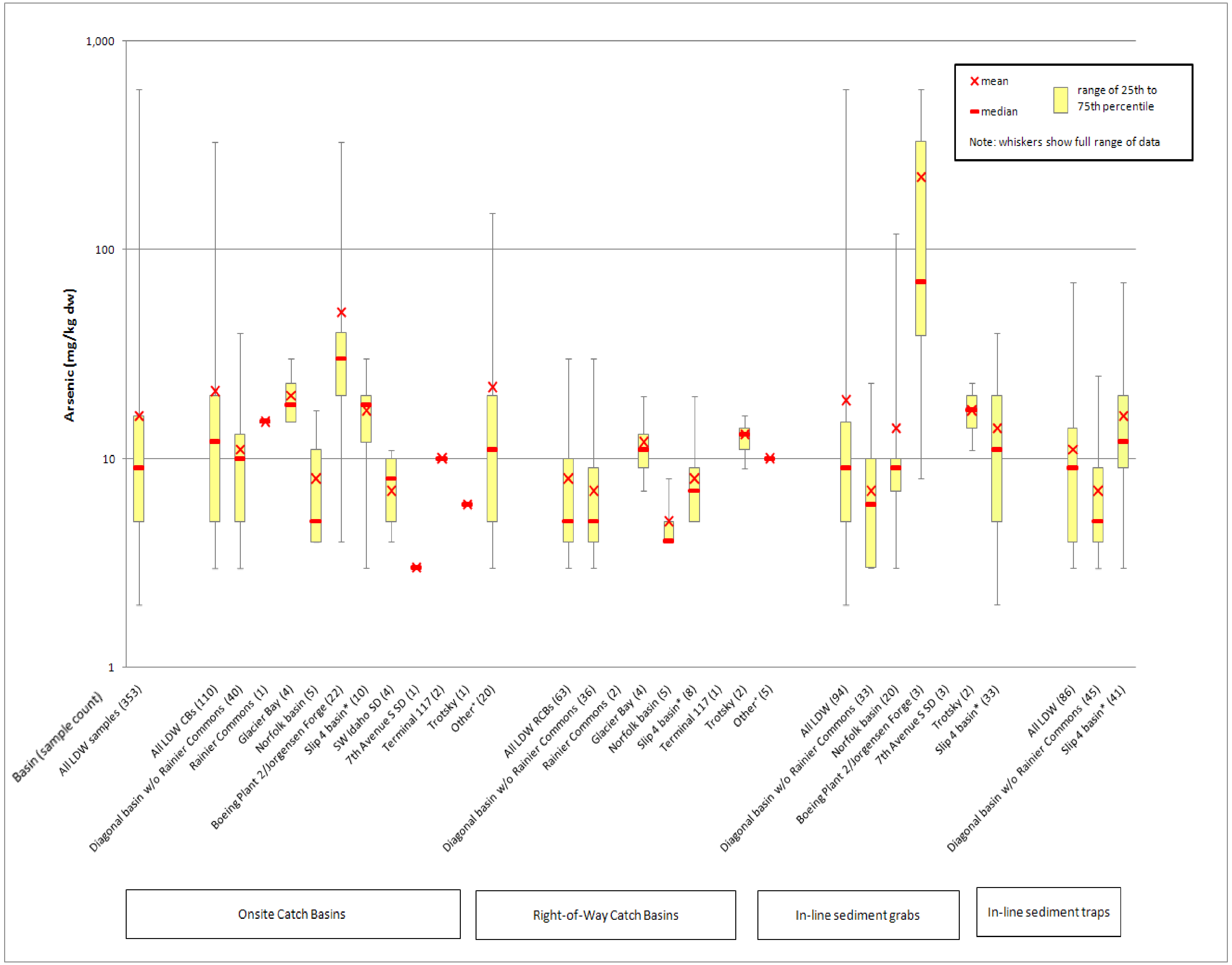
If storm drain sediment samples are below the SQS, it is likely that LDW sediment in outfall discharge areas would not be re-contaminated to levels above the SQS as a result of outfall discharges. In contrast, an exceedance of the SQS in storm drain samples may indicate a potential for sediment recontamination, but additional source investigations would be necessary to determine if recontamination above the SQS might occur. This section summarizes results from the source tracing sampling efforts.

Overall trends and conclusions from the source tracing information collected for all LDW drainages through December 2007 are as follows:

- Chemical concentrations in the four different types of samples were variable.
- Chemicals that frequently had concentrations greater than the CSL included zinc, BEHP, and BBP.

- Other chemicals that had concentrations greater than the SQS in at least one sample included total PCBs, arsenic, copper, lead, mercury, PAHs, di-n-octylphthalate, benzoic acid, 1,2-dichlorobenzene, 1,4-dichlorobenzene, hexachlorobenzene, dibenzofuran, 2-methylnaphthalene, n-nitrosodiphenylamine, and bis(2-chloroethoxy)methane.
- Chemical concentrations were generally higher in samples collected from onsite catch basins than from ROW catch basin, likely because most onsite catch basin sampling stations were selected based on source control problems identified during business inspections.
- Total PCBs were frequently detected in all sample types (70 to 100%, depending on sample type). The majority of higher total PCB concentrations were collected from a few locations in the LDW drainage basin. Elevated total PCB concentrations have generally been associated with specific sources, including PCB-contaminated exterior building paint, paving caulk, or historical activities.
- BEHP and PAHs were detected in most of the samples collected in the LDW drainage basin.
- Arsenic was detected in 38 to 52% of the source tracing samples, depending on sample type; arsenic concentrations were greater than the CSL in six source tracing samples.

Figures 9-5 through 9-10 summarize the results of source tracing samples for arsenic, total PCBs, BBP, BEHP, total LPAHs, and total HPAHs, respectively. General statistical information on chemicals for which source tracing data is available are summarized in Table 9-14. All data have been provided by the City of Seattle (Schmoyer 2008e); an independent QA/QC of the raw data was not performed as part of the RI. Surface sediment data collected near the various drainages referred to in Table 9-14 are discussed in relation to the SQS in Section 4.2.1. Maps 9-12 through 9-14 provide a spatial presentation of source tracing information for total PCBs, mercury, and BEHP concentration ranges, respectively.

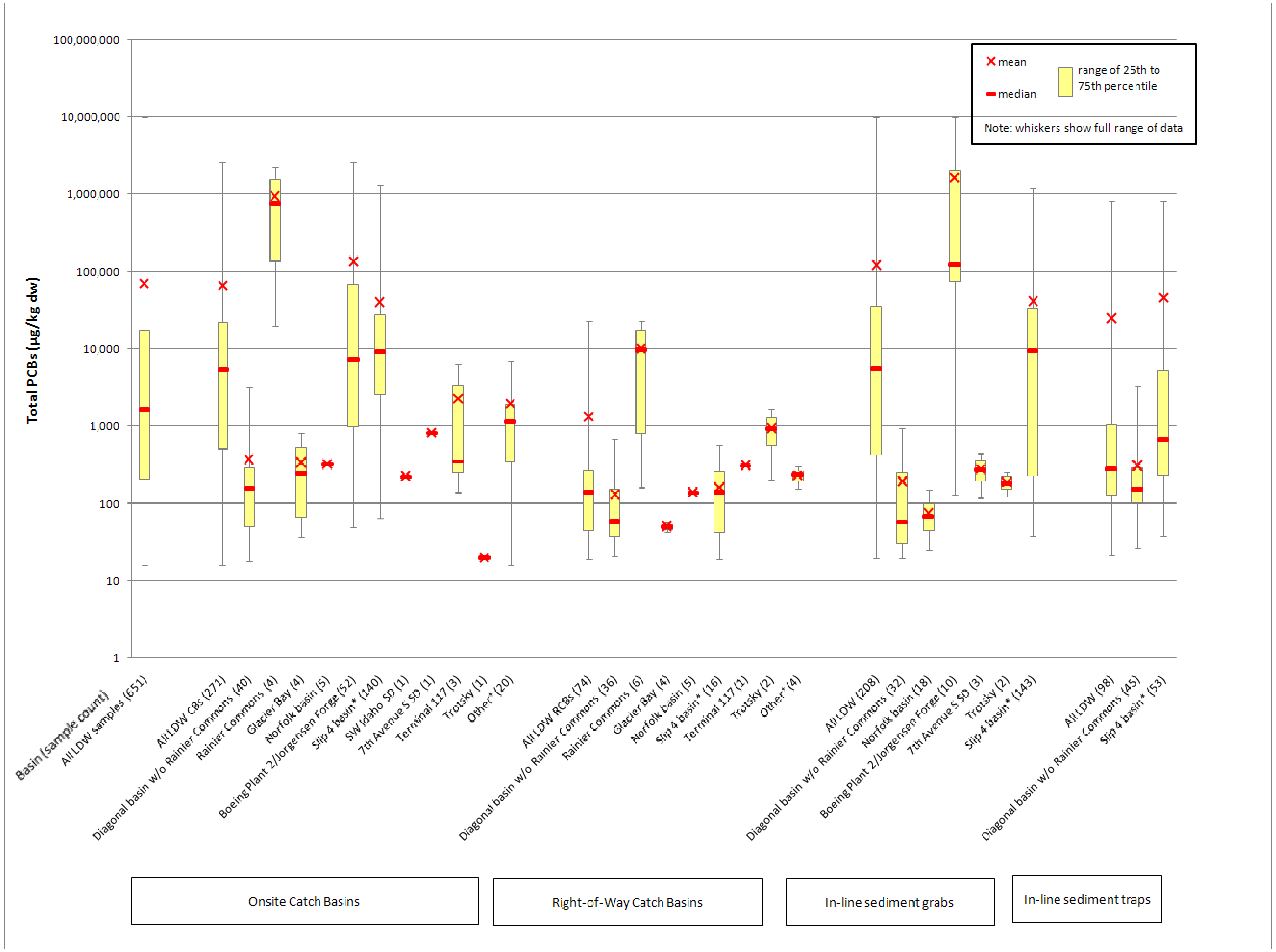


Note: Source tracing data presented in this figure reflect data collected through December 31, 2007 (Schmoyer 2008e). Independent QA/QC of raw data was not performed as part of the RI. LDW source control efforts are ongoing.

* Data from sub-basins within the Slip 4 drainage basin (i.e., Slip 4, NBF, KCIA, I-5, and adjacent properties) have been combined to calculate summary statistics (e.g., ranges, medians). Chemical concentrations within the sub-basins vary greatly. Maps 9-12 through 9-14 provide a spatial presentation of total PCBs, mercury, and BEHP concentrations, respectively.

+ Other samples include source tracing information associated with Slip 5, Slip 6, T-115, and the Chemithon Corporation property.

Figure 9-5. Arsenic source-tracing sample results

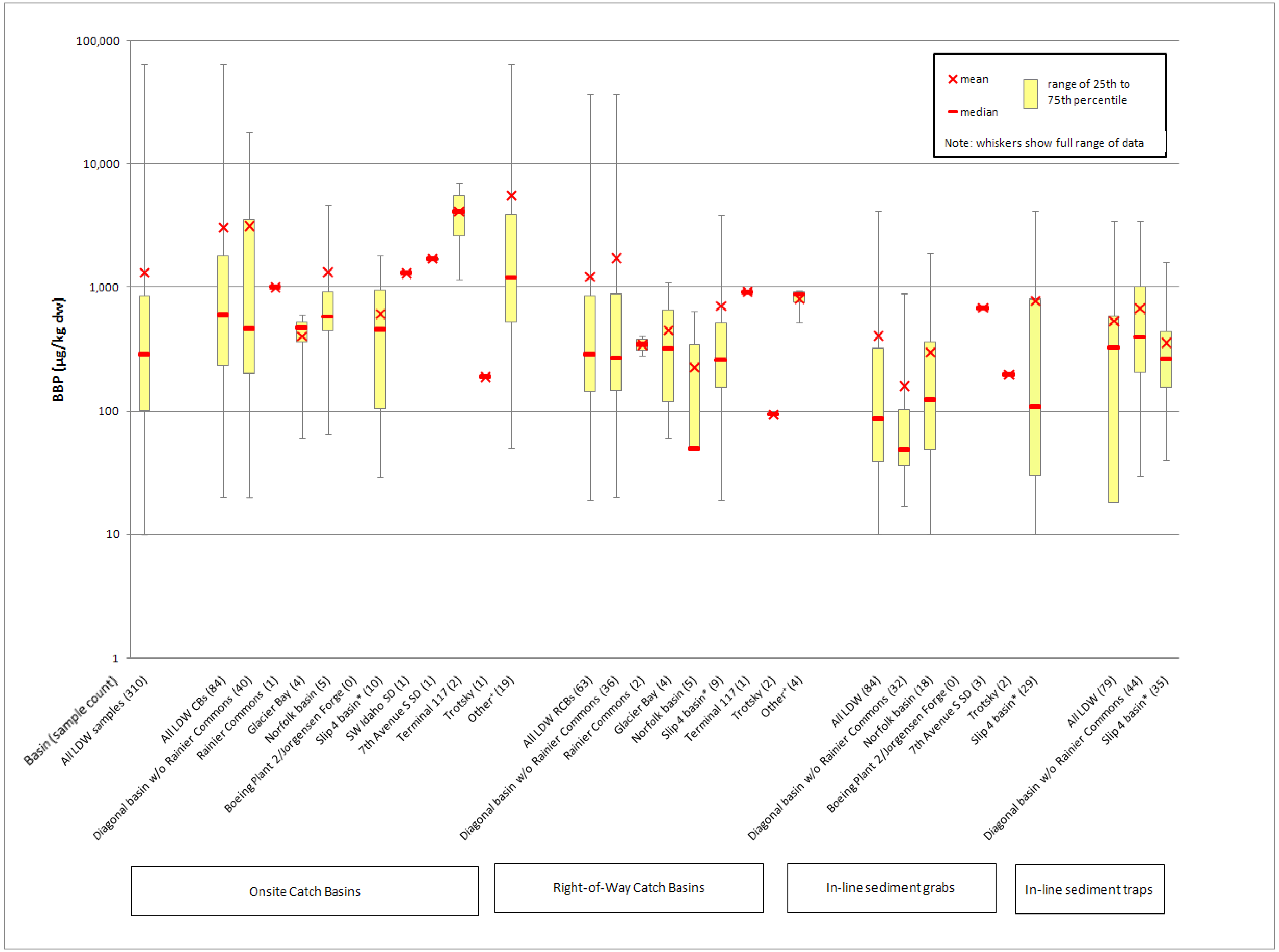


Note: Source tracing data presented in this figure reflect data collected through December 31, 2007 (Schmoyer 2008e). Independent QA/QC of raw data was not performed as part of the RI. LDW source control efforts are ongoing.

* Data from sub-basins within the Slip 4 drainage basin (i.e., Slip 4, NBF, KCIA, I-5, and adjacent properties) have been combined to calculate summary statistics (e.g., ranges, medians). Chemical concentrations within the sub-basins vary greatly. Maps 9-12 through 9-14 provide a spatial presentation of total PCBs, mercury, and BEHP concentrations, respectively.

+ Other samples include source tracing information associated with Slip 5, Slip 6, T-115, and the Chemithon Corporation property.

Figure 9-6. Total PCB source-tracing sample results

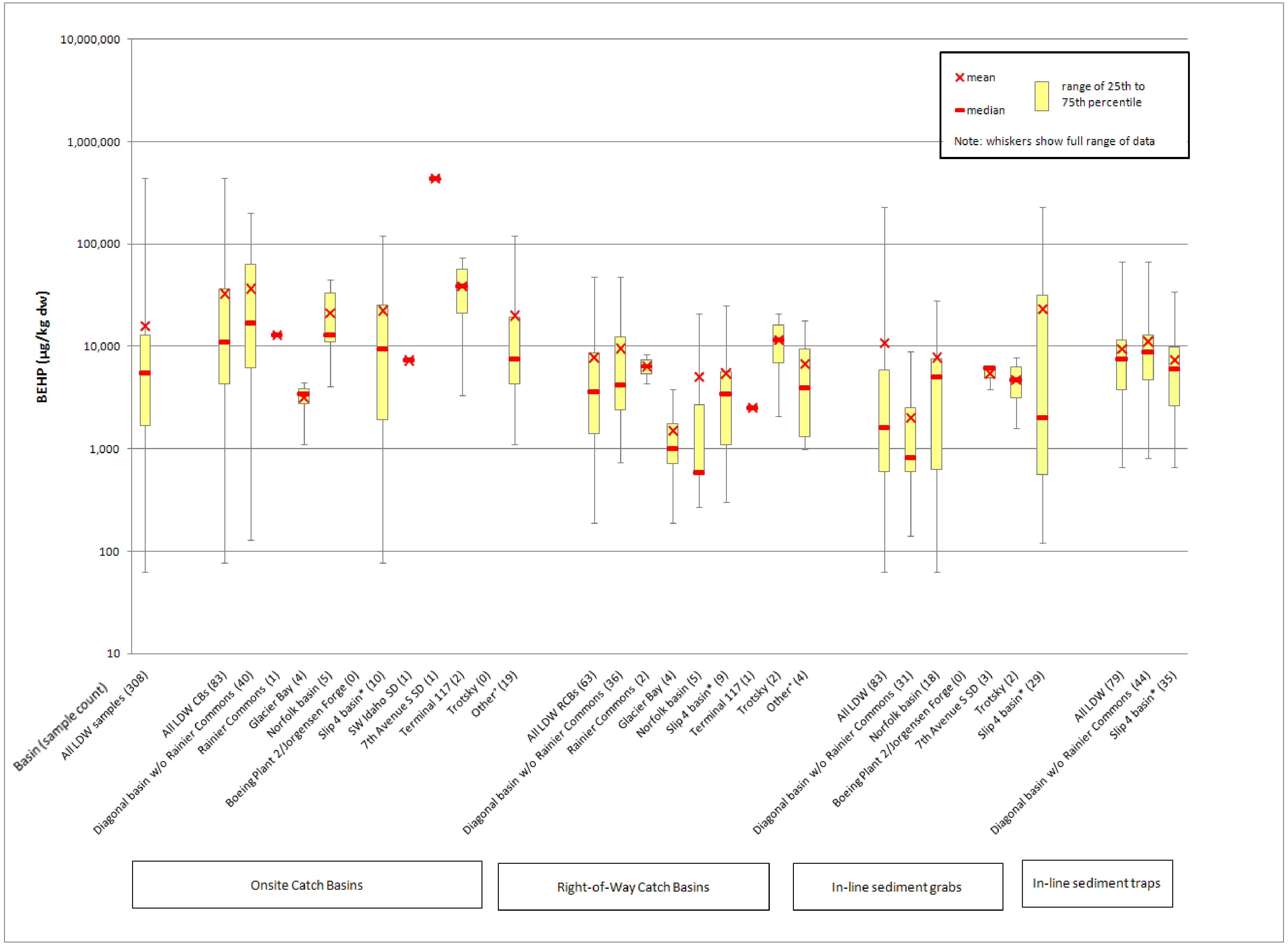


Note: Source tracing data presented in this figure reflect data collected through December 31, 2007 (Schmoyer 2008e). Independent QA/QC of raw data was not performed as part of the RI. LDW source control efforts are ongoing.

* Data from sub-basins within the Slip 4 drainage basin (i.e., Slip 4, NBF, KCIA, I-5, and adjacent properties) have been combined to calculate summary statistics (e.g., ranges, medians). Chemical concentrations within the sub-basins vary greatly. Maps 9-12 through 9-14 provide a spatial presentation of total PCBs, mercury, and BEHP concentrations, respectively.

+ Other samples include source tracing information associated with Slip 5, Slip 6, T-115, and the Chemithon Corporation property.

Figure 9-7. BBP source-tracing sample results

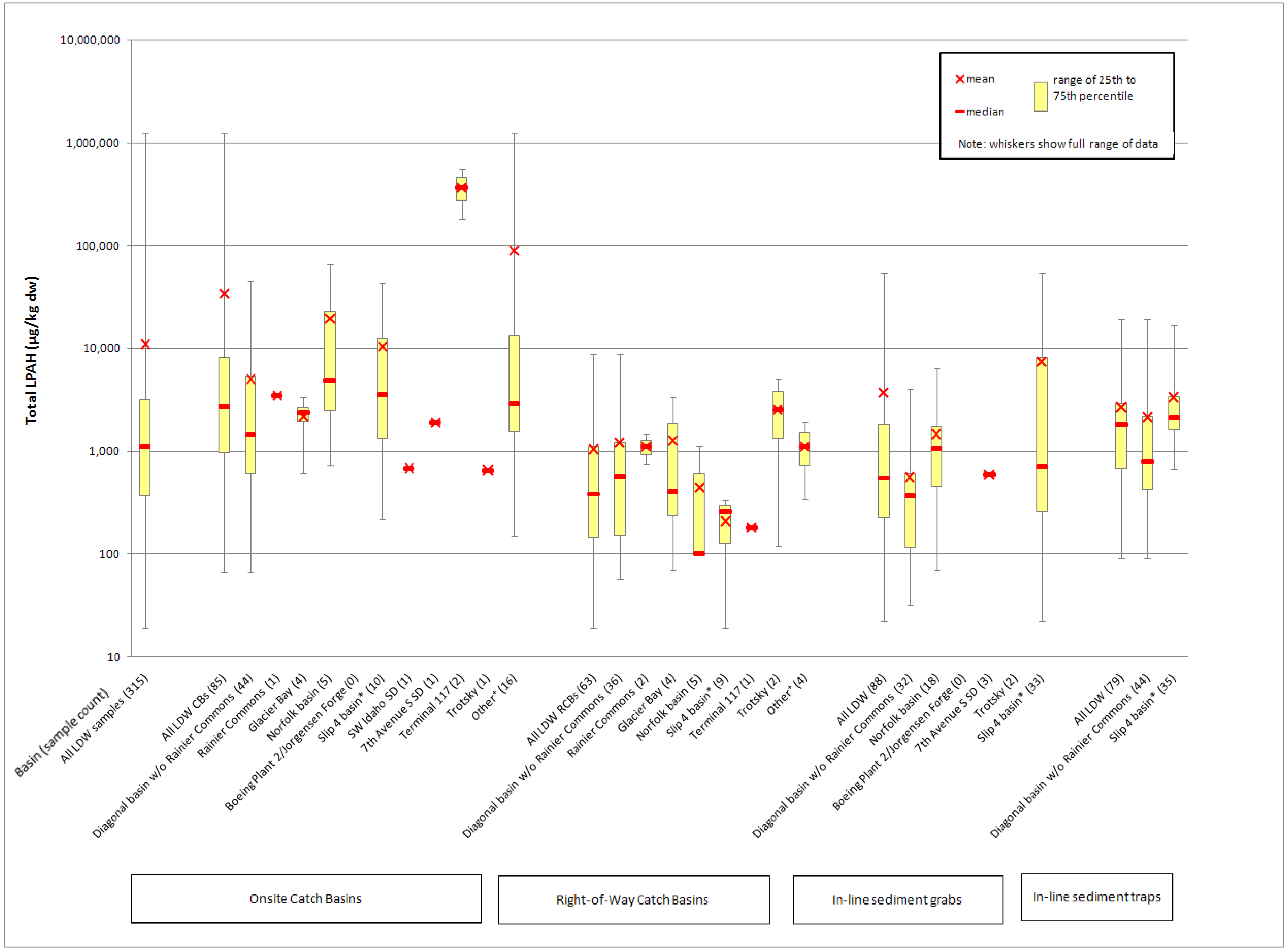


Note: Source tracing data presented in this figure reflect data collected through December 31, 2007 (Schmoyer 2008e). Independent QA/QC of raw data was not performed as part of the RI. LDW source control efforts are ongoing.

* Data from sub-basins within the Slip 4 drainage basin (i.e., Slip 4, NBF, KCIA, I-5, and adjacent properties) have been combined to calculate summary statistics (e.g., ranges, medians). Chemical concentrations within the sub-basins vary greatly. Maps 9-12 through 9-14 provide a spatial presentation of total PCBs, mercury, and BEHP concentrations, respectively.

+ Other samples include source tracing information associated with Slip 5, Slip 6, T-115, and the Chemithon Corporation property.

Figure 9-8. BEHP source-tracing sample results

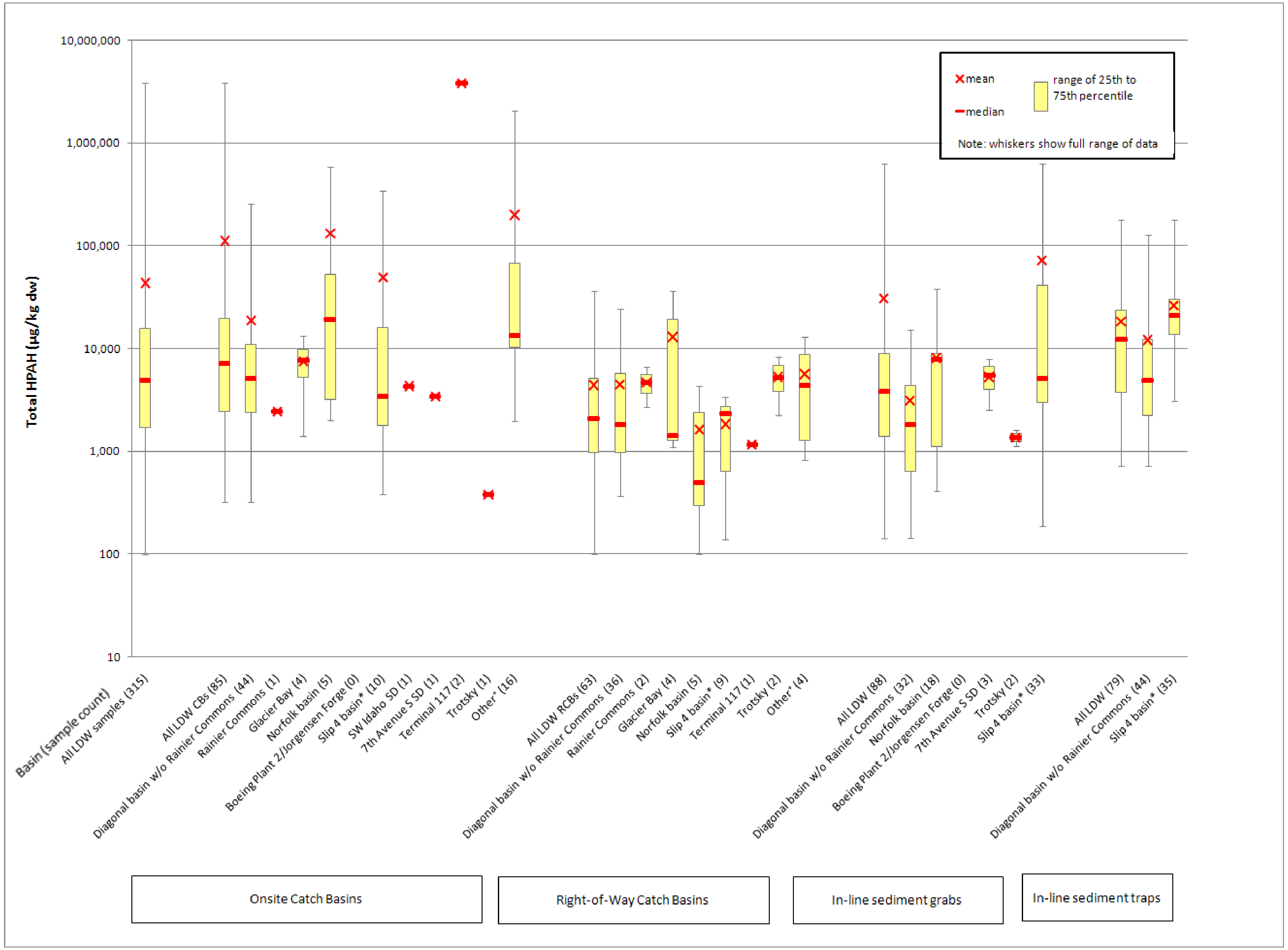


Note: Source tracing data presented in this figure reflect data collected through December 31, 2007 (Schmoyer 2008e). Independent QA/QC of raw data was not performed as part of the RI. LDW source control efforts are ongoing.

* Data from sub-basins within the Slip 4 drainage basin (i.e., Slip 4, NBF, KCIA, I-5, and adjacent properties) have been combined to calculate summary statistics (e.g., ranges, medians). Chemical concentrations within the sub-basins vary greatly. Maps 9-12 through 9-14 provide a spatial presentation of total PCBs, mercury, and BEHP concentrations, respectively.

+ Other samples include source tracing information associated with Slip 5, Slip 6, T-115, and the Chemithon Corporation property.

Figure 9-9. Total LPAH source-tracing sample results



Note: Source tracing data presented in this figure reflect data collected through December 31, 2007 (Schmoyer 2008e). Independent QA/QC of raw data was not performed as part of the RI. LDW source control efforts are ongoing.

* Data from sub-basins within the Slip 4 drainage basin (i.e., Slip 4, NBF, KCIA, I-5, and adjacent properties) have been combined to calculate summary statistics (e.g., ranges, medians). Chemical concentrations within the sub-basins vary greatly. Maps 9-12 through 9-14 provide a spatial presentation of total PCBs, mercury, and BEHP concentrations, respectively.

⁺ Other samples include source tracing information associated with Slip 5, Slip 6, T-115, and the Chemithon Corporation property.

Figure 9-10. HPAH source-tracing sample results

Table 9-14. Summary of data for a subset of chemicals in source tracing samples by sample type and basin

Chemical[a]	Units	Sample Count[b]	Detection Frequency (%)	Range of Detected Concentrations	Percent > SQS and ≤ CSL[c]	Percent > CSL[c]
Onsite Catch Basins						
All LDW Onsite Catch Basins (82 to 271 samples)						
Arsenic	mg/kg dw	110	52%	3 – 330	1%	4%
Copper	mg/kg dw	84	100%	30 – 6,320	21%	0%
Lead	mg/kg dw	110	99%	10 – 15,300	5%	18%
Mercury	mg/kg dw	110	74%	0.02 – 49	3%	23%
Zinc	mg/kg dw	84	100%	55 – 3,940	49%	24%
Total PCBs	µg/kg dw	271	89%	16 – 2,600,000	9%	8%
BBP	µg/kg dw	84	71%	20 – 64,000	50%	13%
BEHP	µg/kg dw	83	100%	77 – 440,000	22%	61%
TPH-Oil	mg/kg dw	82	100%	52 – 390,000	na	na
Total LPAH	µg/kg dw	85	87%	67 – 1,253,000	5%	6%
Total HPAH	µg/kg dw	85	93%	320 – 3,830,000	12%	2%
Diagonal without Rainier Commons Onsite Catch Basins (39 to 44 samples)						
Arsenic	mg/kg dw	40	38%	3 – 40	0%	0%
Copper	mg/kg dw	40	100%	30 – 1,520	13%	0%
Lead	mg/kg dw	40	98%	10 – 5,830	5%	15%
Mercury	mg/kg dw	40	65%	0.02 – 2.05	3%	15%
Zinc	mg/kg dw	40	100%	55 – 3,940	47%	18%
Total PCBs	µg/kg dw	40	73%	18 – 3,200	5%	3%
BBP	µg/kg dw	40	70%	20 – 18,000	45%	15%
BEHP	µg/kg dw	40	100%	130 – 200,000	0%	88%
TPH-Oil	mg/kg dw	39	100%	52 – 250,000	na	na
Total LPAH	µg/kg dw	44	80%	67 – 44,860	2%	0%
Total HPAH	µg/kg dw	44	91%	320 – 256,800	5%	0%
Diagonal with Rainier Commons Onsite Catch Basins (1 to 4 samples)						
Arsenic	mg/kg dw	1	0%	15	0%	0%
Copper	mg/kg dw	1	100%	362	0%	0%
Lead	mg/kg dw	1	100%	430	0%	0%
Mercury	mg/kg dw	1	100%	1.51	0%	100%
Zinc	mg/kg dw	1	100%	1,810	0%	100%
Total PCBs	µg/kg dw	4	100%	19,800 – 2,200,000	0%	100%
BBP	µg/kg dw	1	100%	1,000	100%	0%
BEHP	µg/kg dw	1	100%	13,000	0%	100%
TPH-Oil	mg/kg dw	1	100%	3,400	na	na
Total LPAH	µg/kg dw	1	100%	3,451	0%	0%
Total HPAH	µg/kg dw	1	100%	2,436	0%	0%
Glacier Bay Onsite Catch Basins (4 samples)						
Arsenic	mg/kg dw	4	50%	15 – 30	0%	0%
Copper	mg/kg dw	4	100%	234 – 1,730	75%	0%
Lead	mg/kg dw	4	100%	10 – 120	0%	0%

Table 9-14, cont. Summary of data for a subset of chemicals in source tracing samples by sample type and basin

Chemical[a]	Units	Sample Count[b]	Detection Frequency (%)	Range of Detected Concentrations	Percent > SQS and ≤ CSL[c]	Percent > CSL[c]
Mercury	mg/kg dw	4	75%	0.02 – 0.07	0%	0%
Zinc	mg/kg dw	4	100%	167 – 1,360	50%	25%
Total PCBs	µg/kg dw	4	100%	37 – 810	25%	25%
BBP	µg/kg dw	4	100%	60 – 600	75%	0%
BEHP	µg/kg dw	4	100%	1,100 – 4,500	0%	75%
TPH-Oil	mg/kg dw	4	100%	410 – 2,100	na	na
Total LPAH	µg/kg dw	4	100%	604 – 3,370	0%	0%
Total HPAH	µg/kg dw	4	100%	1,390 – 13,190	25%	0%
Norfolk Basin Onsite Catch Basins (4 to 5 samples)						
Arsenic	mg/kg dw	5	40%	4 – 17	0%	0%
Copper	mg/kg dw	5	100%	44 – 164	0%	0%
Lead	mg/kg dw	5	100%	23 – 295	0%	0%
Mercury	mg/kg dw	5	40%	0.03 – 0.2	0%	0%
Zinc	mg/kg dw	5	100%	121 – 1,980	40%	20%
Total PCBs	µg/kg dw	5	20%	320	0%	0%
BBP	µg/kg dw	5	80%	65 – 4,600	40%	0%
BEHP	µg/kg dw	5	100%	4,100 – 45,000	40%	60%
TPH–Oil	mg/kg dw	4	100%	1,700 – 14,000	na	na
Total LPAH	µg/kg dw	5	100%	730 – 66,230	20%	0%
Total HPAH	µg/kg dw	5	100%	2,010 – 585,400	20%	0%
Boeing Plant 2/Jorgensen Forge Onsite Catch Basins (0 to 52 samples)						
Arsenic	mg/kg dw	22	77%	4 – 330	4%	14%
Copper	mg/kg dw	0	na	na	na	na
Lead	mg/kg dw	22	100%	14 – 15,300	9%	59%
Mercury	mg/kg dw	22	95%	0.03 – 49	9%	68%
Zinc	mg/kg dw	0	na	na	na	na
Total PCBs	µg/kg dw	52	94%	50 – 2,600,000	na[e]	na[e]
BBP	µg/kg dw	0	na	na	na	na
BEHP	µg/kg dw	0	na	na	na	na
TPH-Oil	mg/kg dw	0	na	na	na	na
Total LPAH	µg/kg dw	0	na	na	na	na
Total HPAH	µg/kg dw	0	na	na	na	na
Slip 4 Basin Onsite Catch Basins (10 to 140 samples)[d]						
Arsenic	mg/kg dw	10	80%	3 – 30	0%	0%
Copper	mg/kg dw	10	100%	52 – 6,320	20%	0%
Lead	mg/kg dw	10	100%	29 – 481	10%	0%
Mercury	mg/kg dw	10	90%	0.03 – 0.34	0%	0%
Zinc	mg/kg dw	10	100%	268 – 3,530	60%	30%
Total PCBs	µg/kg dw	140	96%	66 – 1,310,000	10%	0%
BBP	µg/kg dw	10	60%	29 – 1,800	40%	10%
BEHP	µg/kg dw	10	100%	77 – 120,000	0%	50%
TPH-Oil	mg/kg dw	10	100%	210 – 13,000	na	na
Total LPAH	µg/kg dw	10	80%	220 – 43,000	10%	0%
Total HPAH	µg/kg dw	10	100%	380 – 344,400	30%	0%

Table 9-14, cont. Summary of data for a subset of chemicals in source tracing samples by sample type and basin

Chemical[a]	Units	Sample Count[b]	Detection Frequency (%)	Range of Detected Concentrations	Percent > SQS and ≤ CSL[c]	Percent > CSL[c]
SW Idaho SD Onsite Catch Basins (0 to 4 samples)						
Arsenic	mg/kg dw	4	50%	4 – 11	0%	0%
Copper	mg/kg dw	0	na	na	na	na
Lead	mg/kg dw	4	100%	36 – 127	0%	0%
Mercury	mg/kg dw	4	50%	0.03 – 0.13	0%	0%
Zinc	mg/kg dw	0	na	na	na	na
Total PCBs	µg/kg dw	1	100%	223	0%	0%
BBP	µg/kg dw	1	100%	1,300	100%	0%
BEHP	µg/kg dw	1	100%	7,300	0%	100%
TPH-Oil	mg/kg dw	1	100%	1,700	na	na
Total LPAH	µg/kg dw	1	100%	680	0%	0%
Total HPAH	µg/kg dw	1	100%	4,310	0%	0%
7th Avenue S SD Onsite Catch Basins (1 sample)						
Arsenic	mg/kg dw	1	0%	na	0%	0%
Copper	mg/kg dw	1	100%	285	0%	0%
Lead	mg/kg dw	1	100%	40	0%	0%
Mercury	mg/kg dw	1	100%	0.05	0%	0%
Zinc	mg/kg dw	1	100%	392	0%	0%
Total PCBs	µg/kg dw	1	100%	810	0%	0%
BBP	µg/kg dw	1	100%	1,700	0%	0%
BEHP	µg/kg dw	1	100%	440,000	0%	100%
TPH-Oil	mg/kg dw	1	100%	390,000	na	na
Total LPAH	µg/kg dw	1	100%	1,900	0%	0%
Total HPAH	µg/kg dw	1	100%	3,400	0%	0%
Terminal 117 Onsite Catch Basins (2 to 3 samples)						
Arsenic	mg/kg dw	2	0%	10	0%	0%
Copper	mg/kg dw	2	100%	134 – 173	0%	0%
Lead	mg/kg dw	2	100%	98 – 428	0%	0%
Mercury	mg/kg dw	2	50%	0.04 – 0.11	0%	0%
Zinc	mg/kg dw	2	100%	711 – 830	100%	0%
Total PCBs	µg/kg dw	3	100%	140 – 6300	0%	0%
BBP	µg/kg dw	2	0%	1,150 – 7,000	100%	0%
BEHP	µg/kg dw	2	100%	3,300 – 74,000	0%	50%
TPH-Oil	mg/kg dw	2	100%	19,000 – 77,000	na	na
Total LPAH	µg/kg dw	2	100%	183,000 – 556,000	0%	100%
Total HPAH	µg/kg dw	2	50%	3,830,000	0%	50%
Trotsky Onsite Catch Basins (0 to 1 samples)						
Arsenic	mg/kg dw	1	0%	6	0%	0%
Copper	mg/kg dw	1	100%	146	0%	0%
Lead	mg/kg dw	1	100%	25	0%	0%
Mercury	mg/kg dw	1	0%	na	0%	0%
Zinc	mg/kg dw	1	100%	111	0%	0%
Total PCBs	µg/kg dw	1	100%	20	0%	0%
BBP	µg/kg dw	1	100%	190	100%	0%

Table 9-14, cont. Summary of data for a subset of chemicals in source tracing samples by sample type and basin

Chemical[a]	Units	Sample Count[b]	Detection Frequency (%)	Range of Detected Concentrations	Percent > SQS and ≤ CSL[c]	Percent > CSL[c]
BEHP	µg/kg dw	0	na	na	na	na
TPH-Oil	mg/kg dw	1	100%	4,600	na	na
Total LPAH	µg/kg dw	1	100%	650	0%	0%
Total HPAH	µg/kg dw	1	100%	380	0%	0%
Other Onsite Catch Basins (16 to 20 samples)[f]						
Arsenic	mg/kg dw	20	55%	3 – 150	5%	5%
Copper	mg/kg dw	20	100%	79 – 1,820	40%	0%
Lead	mg/kg dw	20	100%	47 – 1,760	0%	5%
Mercury	mg/kg dw	20	75%	0.03 – 9.4	5%	15%
Zinc	mg/kg dw	20	100%	271 – 3,290	50%	35%
Total PCBs	µg/kg dw	20	60%	16 – 7000	16%	21%
BBP	µg/kg dw	19	74%	50 – 64,000	53%	21%
BEHP	µg/kg dw	19	95%	1,100 – 120,000	5%	79%
TPH–Oil	mg/kg dw	19	100%	650 – 25,000	na	na
Total LPAH	µg/kg dw	16	100%	150 – 1,253,000	7%	20%
Total HPAH	µg/kg dw	16	94%	1,952 – 2,062,000	20%	7%
ROW Catch Basins						
All LDW ROW Catch Basins (62 to 73 samples)						
Arsenic	mg/kg dw	63	38%	3 – 30	0%	0%
Copper	mg/kg dw	62	100%	30 – 751	3%	0%
Lead	mg/kg dw	63	100%	10 – 1,370	0%	3%
Mercury	mg/kg dw	63	44%	0.02 – 1.46	0%	5%
Zinc	mg/kg dw	62	100%	73 – 966	27%	2%
Total PCBs	µg/kg dw	73	70%	19 – 23,000	2%	1%
BBP	µg/kg dw	63	75%	19 – 37,000	48%	3%
BEHP	µg/kg dw	63	98%	190 – 48,000	16%	43%
TPH-Oil	mg/kg dw	63	100%	360 – 18,000	na	na
Total LPAH	µg/kg dw	63	87%	19 – 8,900	0%	0%
Total HPAH	µg/kg dw	63	92%	99 – 36,520	0%	0%
Diagonal without Rainier Commons ROW Catch Basins (36 samples)						
Arsenic	mg/kg dw	36	31%	3 – 30	0%	0%
Copper	mg/kg dw	36	100%	38 – 751	3%	0%
Lead	mg/kg dw	36	100%	19 – 1,370	0%	3%
Mercury	mg/kg dw	36	44%	0.02 – 1.17	0%	6%
Zinc	mg/kg dw	36	100%	85 – 966	25%	3%
Total PCBs	µg/kg dw	36	61%	21 – 670	0%	0%
BBP	µg/kg dw	36	78%	20 – 37,000	41%	6%
BEHP	µg/kg dw	36	100%	740 – 48,000	8%	56%
TPH–Oil	mg/kg dw	36	100%	480 – 14,000	na	na
Total LPAH	µg/kg dw	36	92%	57 – 8,900	0%	0%
Total HPAH	µg/kg dw	36	94%	364 – 24,290	0%	0%
Diagonal with Rainier Commons ROW Catch Basins (2 to 6 samples)						
Arsenic	mg/kg dw	2	0%	na	0%	0%
Copper	mg/kg dw	2	100%	59 – 113	0%	0%

Table 9-14, cont. Summary of data for a subset of chemicals in source tracing samples by sample type and basin

Chemical[a]	Units	Sample Count[b]	Detection Frequency (%)	Range of Detected Concentrations	Percent > SQS and ≤ CSL[c]	Percent > CSL[c]
Lead	mg/kg dw	2	100%	61 – 62	0%	0%
Mercury	mg/kg dw	2	0%	na	0%	0%
Zinc	mg/kg dw	2	100%	189 – 213	0%	0%
Total PCBs	µg/kg dw	6	100%	160 – 23,000	0%	50%
BBP	µg/kg dw	2	100%	280 – 410	100%	0%
BEHP	µg/kg dw	2	100%	4,400 – 8,300	0%	100%
TPH-Oil	mg/kg dw	2	100%	1,200 – 3,500	na	na
Total LPAH	µg/kg dw	2	100%	750 – 1,448	0%	0%
Total HPAH	µg/kg dw	2	100%	2693 – 6,600	0%	0%
Glacier Bay ROW Catch Basins (4 samples)						
Arsenic	mg/kg dw	4	100%	7 – 20	0%	0%
Copper	mg/kg dw	4	100%	36 – 183	0%	0%
Lead	mg/kg dw	4	100%	11 – 402	0%	0%
Mercury	mg/kg dw	4	25%	0.09	0%	0%
Zinc	mg/kg dw	4	100%	78 – 635	25%	0%
Total PCBs	µg/kg dw	4	50%	43 – 58	0%	0%
BBP	µg/kg dw	4	75%	60 – 1,100	75%	0%
BEHP	µg/kg dw	4	100%	190 – 3,800	75%	0%
TPH-Oil	mg/kg dw	4	100%	370 – 1,700	na	na
Total LPAH	µg/kg dw	4	75%	69 – 3,320	0%	0%
Total HPAH	µg/kg dw	4	75%	1100 – 36,520	0%	0%
Norfolk Basin ROW Catch Basins (5 samples)						
Arsenic	mg/kg dw	5	20%	4 – 8	0%	0%
Copper	mg/kg dw	5	100%	29 – 98	0%	0%
Lead	mg/kg dw	5	100%	20 – 126	0%	0%
Mercury	mg/kg dw	5	0%	0.03 – 0.05	0%	0%
Zinc	mg/kg dw	5	100%	73 – 305	0%	0%
Total PCBs	µg/kg dw	5	20%	138	0%	0%
BBP	µg/kg dw	5	40%	50 – 640	20%	0%
BEHP	µg/kg dw	5	100%	270 – 21,000	0%	20%
TPH-Oil	mg/kg dw	5	100%	360 – 1,500	na	na
Total LPAH	µg/kg dw	5	60%	99 – 1,120	0%	0%
Total HPAH	µg/kg dw	5	100%	99 – 4,300	0%	0%
Slip 4 Basin ROW Catch Basins (8 to 16 samples)[d]						
Arsenic	mg/kg dw	8	63%	5 – 20	0%	0%
Copper	mg/kg dw	10	100%	38 – 720	10%	0%
Lead	mg/kg dw	9	100%	10 – 179	0%	0%
Mercury	mg/kg dw	9	67%	0.02 – 0.11	0%	0%
Zinc	mg/kg dw	9	100%	93 – 593	44%	0%
Total PCBs	µg/kg dw	16	100%	19 – 570	0%	0%
BBP	µg/kg dw	9	78%	19 – 3,800	56%	0%
BEHP	µg/kg dw	9	100%	300 – 25,000	34%	22%
TPH-Oil	mg/kg dw	9	100%	400 – 8,000	na	na
Total LPAH	µg/kg dw	9	89%	19 – 330	0%	0%

Table 9-14, cont. Summary of data for a subset of chemicals in source tracing samples by sample type and basin

Chemical[a]	Units	Sample Count[b]	Detection Frequency (%)	Range of Detected Concentrations	Percent > SQS and ≤ CSL[c]	Percent > CSL[c]
Total HPAH	µg/kg dw	9	100%	140 – 3,380	0%	0%
Terminal 117 ROW Catch Basins (1 sample)						
Arsenic	mg/kg dw	1	0%	na	0%	0%
Copper	mg/kg dw	1	100%	51	0%	0%
Lead	mg/kg dw	1	100%	22	0%	0%
Mercury	mg/kg dw	1	0%	na	0%	0%
Zinc	mg/kg dw	1	100%	237	0%	0%
Total PCBs	µg/kg dw	1	100%	310	0%	0%
BBP	µg/kg dw	1	100%	920	100%	0%
BEHP	µg/kg dw	1	100%	2,500	0%	0%
TPH-Oil	mg/kg dw	1	100%	4,100	na	na
Total LPAH	µg/kg dw	1	100%	180	0%	0%
Total HPAH	µg/kg dw	1	100%	1,160	0%	0%
Trotsky ROW Catch Basins (2 samples)						
Arsenic	mg/kg dw	2	100%	9 – 16	0%	0%
Copper	mg/kg dw	2	100%	141 – 254	0%	0%
Lead	mg/kg dw	2	100%	115 – 547	0%	50%
Mercury	mg/kg dw	2	100%	0.13 – 1.46	0%	50%
Zinc	mg/kg dw	2	100%	592 – 655	100%	0%
Total PCBs	µg/kg dw	2	100%	203 – 1650	50%	0%
BBP	µg/kg dw	2	50%	95	0%	0%
BEHP	µg/kg dw	2	100%	2,100 – 21,000	0%	50%
TPH-Oil	mg/kg dw	2	100%	4,600 – 18,000	na	na
Total LPAH	µg/kg dw	2	100%	120 – 4,990	0%	0%
Total HPAH	µg/kg dw	2	100%	2,266 – 8,320	0%	0%
Other ROW Catch Basins (2 to 5 samples)[f]						
Arsenic	mg/kg dw	5	20%	10	0%	0%
Copper	mg/kg dw	2	100%	85 – 157	0%	0%
Lead	mg/kg dw	4	100%	70 – 145	0%	0%
Mercury	mg/kg dw	4	75%	0.03 – 0.2	0%	0%
Zinc	mg/kg dw	3	100%	329 – 781	33%	0%
Total PCBs	µg/kg dw	4	50%	158 – 303	0%	0%
BBP	µg/kg dw	4	75%	520 – 950	75%	0%
BEHP	µg/kg dw	4	75%	1,000 – 18,000	25%	25%
TPH-Oil	mg/kg dw	4	100%	450 – 9,100	na	na
Total LPAH	µg/kg dw	4	75%	340 – 1,900	0%	0%
Total HPAH	µg/kg dw	4	100%	820 – 13,000	0%	0%
In-Line Sediment Grabs						
All LDW In-Line Sediment Grabs (81 to 208 samples)						
Arsenic	mg/kg dw	94	52%	2 – 590	1%	2%
Copper	mg/kg dw	91	100%	13 – 3,960	3%	0%
Lead	mg/kg dw	94	100%	10 – 47,200	2%	9%
Mercury	mg/kg dw	94	53%	0.01 – 3.3	2%	10%
Zinc	mg/kg dw	87	100%	40 – 9,980	26%	13%

Table 9-14, cont. **Summary of data for a subset of chemicals in source tracing samples by sample type and basin**

CHEMICAL[a]	UNITS	SAMPLE COUNT[b]	DETECTION FREQUENCY (%)	RANGE OF DETECTED CONCENTRATIONS	PERCENT > SQS AND ≤ CSL[c]	PERCENT > CSL[c]
Total PCBs	µg/kg dw	208	86%	20 – 10,000,000	10%	10%
BBP	µg/kg dw	84	40%	10 – 4,090	30%	1%
BEHP	µg/kg dw	83	100%	63 – 232,000	19%	47%
TPH-Oil	mg/kg dw	81	100%	61 – 81,000	na	na
Total LPAH	µg/kg dw	88	84%	22 – 53,560	5%	1%
Total HPAH	µg/kg dw	88	98%	143 – 629,900	13%	2%
Diagonal Basin In-Line Sediment Grabs (31 to 33 samples)						
Arsenic	mg/kg dw	33	12%	3 – 23	0%	0%
Copper	mg/kg dw	33	100%	22 – 340	0%	0%
Lead	mg/kg dw	33	100%	15 – 4,910	0%	9%
Mercury	mg/kg dw	33	18%	0.01 – 3.3	0%	9%
Zinc	mg/kg dw	33	100%	85 – 718	18%	0%
Total PCBs	µg/kg dw	32	44%	20 – 940	3%	0%
BBP	µg/kg dw	32	34%	17 – 900	25%	0%
BEHP	µg/kg dw	31	100%	140 – 8,900	25%	52%
TPH-Oil	mg/kg dw	32	100%	110 – 13,000	na	na
Total LPAH	µg/kg dw	32	84%	32 – 4,000	0%	0%
Total HPAH	µg/kg dw	32	97%	143 – 15,300	6%	0%
Norfolk Basin In-Line Sediment Grabs (16 to 20 samples)						
Arsenic	mg/kg dw	20	70%	3 – 120	0%	5%
Copper	mg/kg dw	20	100%	20 – 3,960	5%	0%
Lead	mg/kg dw	20	100%	16 – 700	0%	5%
Mercury	mg/kg dw	20	55%	0.02 – 0.4	0%	0%
Zinc	mg/kg dw	20	100%	90 – 9,980	35%	15%
Total PCBs	µg/kg dw	18	78%	25 – 150	0%	0%
BBP	µg/kg dw	18	33%	10 – 1,900	33%	0%
BEHP	µg/kg dw	18	100%	63 – 28,000	0%	56%
TPH-Oil	mg/kg dw	16	100%	180 – 5,300	na	na
Total LPAH	µg/kg dw	18	78%	70 – 6,390	0%	0%
Total HPAH	µg/kg dw	18	94%	410 – 37,790	0%	0%
Boeing Plant 2/Jorgensen Forge In-Line Sediment Grabs (0 to 10 samples)						
Arsenic	mg/kg dw	3	100%	8 – 590	34%	33%
Copper	mg/kg dw	0	na	na	na	na
Lead	mg/kg dw	3	100%	52 – 47,200	0%	67%
Mercury	mg/kg dw	3	100%	1.37 – 3.24	0%	100%
Zinc	mg/kg dw	0	na	na	na	na
Total PCBs	µg/kg dw	10	100%	130 – 100,00,000	nc[e]	nc[e]
BBP	µg/kg dw	0	na	na	na	na
BEHP	µg/kg dw	0	na	na	na	na
TPH-Oil	mg/kg dw	0	na	na	na	na
Total LPAH	µg/kg dw	0	na	na	na	na
Total HPAH	µg/kg dw	0	na	na	na	na
7th Avenue S SD In-Line Sediment Grabs (3 samples)						
Arsenic	mg/kg dw	3	100%	20 – 30	0%	0%

Table 9-14, cont. Summary of data for a subset of chemicals in source tracing samples by sample type and basin

Chemical[a]	Units	Sample Count[b]	Detection Frequency (%)	Range of Detected Concentrations	Percent > SQS and ≤ CSL[c]	Percent > CSL[c]
Copper	mg/kg dw	3	100%	129 – 175	0%	0%
Lead	mg/kg dw	3	100%	119 - 151	0%	0%
Mercury	mg/kg dw	3	100%	0.17 - 0.2	0%	0%
Zinc	mg/kg dw	3	100%	515 - 575	100%	0%
Total PCBs	µg/kg dw	3	100%	119 - 440	0%	0%
BBP	µg/kg dw	3	33%	680	33%	0%
BEHP	µg/kg dw	3	100%	3,800 – 6,400	0%	67%
TPH-Oil	mg/kg dw	3	100%	1,900 – 3,100	na	na
Total LPAH	µg/kg dw	3	33%	590	0%	0%
Total HPAH	µg/kg dw	3	100%	2,530 - 7,880	0%	0%
Trotsky In-Line Sediment Grabs (2 samples)						
Arsenic	mg/kg dw	2	100%	11 - 23	0%	0%
Copper	mg/kg dw	2	100%	99 - 105	0%	0%
Lead	mg/kg dw	2	100%	87 - 113	0%	0%
Mercury	mg/kg dw	2	50%	0.06	0%	0%
Zinc	mg/kg dw	2	100%	394 - 444	50%	0%
Total PCBs	µg/kg dw	2	100%	122 - 250	0%	0%
BBP	µg/kg dw	2	50%	200	0%	0%
BEHP	µg/kg dw	2	100%	1,600 – 7,800	0%	50%
TPH-Oil	mg/kg dw	2	100%	3,100 – 3,900	na	na
Total LPAH	µg/kg dw	2	0%	na	0%	0%
Total HPAH	µg/kg dw	2	100%	1,130 – 1,600	0%	0%
Slip 4 Basin In-Line Sediment Grabs (28 to 143 samples)[d]						
Arsenic	mg/kg dw	33	70%	2 – 40	0%	0%
Copper	mg/kg dw	33	100%	13 – 1,550	6%	0%
Lead	mg/kg dw	33	100%	10 – 744	6%	6%
Mercury	mg/kg dw	33	79%	0.02 – 1.7	6%	9%
Zinc	mg/kg dw	29	100%	40 – 1,880	20%	28%
Total PCBs	µg/kg dw	143	95%	38 – 1,200,000	21%	27%
BBP	µg/kg dw	29	52%	10 – 4,090	31%	3%
BEHP	µg/kg dw	29	100%	120 – 232,000	28%	34%
TPH-Oil	mg/kg dw	28	100%	61 – 81,000	na	na
Total LPAH	µg/kg dw	33	100%	22 – 53,560	12%	3%
Total HPAH	µg/kg dw	33	100%	185 – 629,900	27%	6%
In-Line Sediment Traps						
All LDW In-Line Sediment Traps (68 to 98 samples)						
Arsenic	mg/kg dw	86	40%	3 – 70	1%	0%
Copper	mg/kg dw	86	100%	4 – 640	5%	0%
Lead	mg/kg dw	86	100%	4 – 1,070	1%	5%
Mercury	mg/kg dw	86	71%	0.02 – 8.3	1%	20%
Zinc	mg/kg dw	86	100%	30 – 2,460	45%	12%
Total PCBs	µg/kg dw	98	96%	22 – 800,000	14%	5%
BBP	µg/kg dw	79	70%	30 – 3,400	41%	0%
BEHP	µg/kg dw	79	100%	670 – 67,000	13%	63%

Table 9-14, cont. Summary of data for a subset of chemicals in source tracing samples by sample type and basin

CHEMICAL[a]	UNITS	SAMPLE COUNT[b]	DETECTION FREQUENCY (%)	RANGE OF DETECTED CONCENTRATIONS	PERCENT > SQS AND ≤ CSL[c]	PERCENT > CSL[c]
TPH-Oil	mg/kg dw	68	100%	120 – 7,500	na	na
Total LPAH	µg/kg dw	79	97%	90 – 19,350	1%	0%
Total HPAH	µg/kg dw	79	100%	720 – 180,300	3%	0%
Diagonal Basin In-Line Sediment Traps (42 to 45 samples)						
Arsenic	mg/kg dw	45	22%	3 – 25	0%	0%
Copper	mg/kg dw	45	100%	7 – 597	4%	0%
Lead	mg/kg dw	45	100%	29 – 360	0%	0%
Mercury	mg/kg dw	45	62%	0.03 – 2.8	0%	4%
Zinc	mg/kg dw	45	100%	162 – 1,930	42%	7%
Total PCBs	µg/kg dw	45	96%	27 – 3,250	0%	2%
BBP	µg/kg dw	44	80%	30 – 3,400	48%	0%
BEHP	µg/kg dw	44	100%	820 – 67,000	13%	73%
TPH-Oil	mg/kg dw	42	100%	120 – 7,500	na	na
Total LPAH	µg/kg dw	44	95%	90 – 19,350	0%	0%
Total HPAH	µg/kg dw	44	100%	720 – 127,580	2%	0%
Slip 4 Basin In-Line Sediment Traps (26 to 53 samples)[d]						
Arsenic	mg/kg dw	41	59%	3 – 70	2%	0%
Copper	mg/kg dw	41	100%	4 – 640	5%	0%
Lead	mg/kg dw	41	100%	4 – 1,070	2%	10%
Mercury	mg/kg dw	41	80%	0.02 – 8.3	2%	37%
Zinc	mg/kg dw	41	100%	30 – 2,460	49%	17%
Total PCBs	µg/kg dw	53	96%	38 – 800,000	15%	33%
BBP	µg/kg dw	35	57%	40 – 1,600	31%	0%
BEHP	µg/kg dw	35	100%	670 – 34,000	12%	51%
TPH–Oil	mg/kg dw	26	100%	410 – 7,500	na	na
Total LPAH	µg/kg dw	35	100%	668 – 16,700	4%	0%
Total HPAH	µg/kg dw	35	77%	3,084 – 180,300	4%	0%

Note: Source tracing data presented in this table reflect data collected through December 31, 2007 (Schmoyer 2008e). Independent QA/QC of raw data was not performed as part of the RI LDW source control efforts are on-going.

[a] Total PCBs, total LPAH, and total HPAH concentrations were calculated using detected concentrations only.

[b] Number of samples per sample type may vary because of insufficient sample volume (i.e., not all samples per sample type were analyzed for all listed chemicals).

[c] For organic compounds with OC-normalized SMS criteria, percentages of samples with concentrations above the SQS and the CSL were based on the number of detected samples for which TOC information was available. Detection frequencies were based on the total number of samples collected in each sub-basin per media type.

[d] Data from sub-basins within the Slip 4 drainage basin (i.e., Slip 4, NBF, KCIA, I-5, and adjacent properties) have been combined to calculate summary statistics (e.g., ranges, medians). Chemical concentrations within the sub-basins vary greatly. Maps 9-12 through 9-14 provide a spatial presentation of total PCBs, mercury, and BEHP concentrations, respectively.

[e] TOC information not available for comparison with SMS.

[f] Other samples include those collected in association with Slip 5, Slip 6, T-115, and the Chemithon corporation property.

BEHP – bis(2-ethylhexyl) phthalate
BBP – butyl benzyl phthalate
CSL – cleanup screening level
HPAH – high molecular weight polycyclic aromatic hydrocarbon
LDW – Lower Duwamish Waterway
LPAH – low molecular weight polycyclic aromatic hydrocarbon
na – not applicable
nc – not calculated
NBF – North Boeing Field
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
ROW – right-of-way
SD – storm drain
SQS – sediment quality standards
T-115 – Terminal 115

Catch Basin Cleaning and Performance Results

In addition to sampling catch basins, the City of Seattle has a maintenance program that routinely cleans catch basins in the public ROWs. Street dirt can contain chemicals such as metals, petroleum hydrocarbons, and other organic chemicals associated with motor vehicles. These chemicals often adsorb onto particulate material and are transported along with suspended solids present in stormwater runoff.

Although catch basins are primarily designed to prevent storm drains from becoming plugged by trapping large debris and solids, some of the particulate-bound chemicals can also be removed in catch basins. At the start of the LDW source control effort, SPU prioritized cleaning of catch basins in the Diagonal Avenue S CSO/SD drainage basin (shown in Map 9-1). In 2006, SPU cleaned catch basins and jetted the storm drain lines located in the public ROW in the 7th Avenue S storm drain system located in the South Park neighborhood (see Maps 9-1 and 9-10). SPU cleaned all the ROW catch basins in the Diagonal Avenue S CSO/SD storm drain basin between August 2007 and March 2008.

Diagonal Avenue S CSO/SD Cleanout Program Performance

In 2002, SPU collected in-line sediment samples from the Diagonal Avenue S CSO/SD as part of an agreement with the Elliott Bay Duwamish Restoration Panel to characterize sediment discharged from this system to the LDW. Sampling was conducted in the lower part of the drainage system, below 4th Avenue S (Map 9-15). Concentrations of lead, mercury, zinc, BEHP, BBP, and PAHs in sediment collected from the main-line on Diagonal Avenue S were greater than the SQS in one or more samples. Concentrations of lead, mercury, zinc, BEHP, BBP, and PCBs in sediment collected from the drainage laterals of the system were greater than the SQS in one or more samples.

From 2002 to 2004, the City of Seattle cleaned portions of the Diagonal Avenue S CSO/SD main-line as well as several of the lateral lines. Approximately 6,200 linear feet (LF) of pipe were cleaned and an estimated 300 cubic yards (cy) of sediments were removed. Since the cleaning was completed, SPU has collected four in-line sediment grabs and six in-line sediment trap samples from a station located on the main-line at the lower end of the drainage system (location M1 on Map 9-15).

In the in-line sediment grab samples, BEHP concentrations were lower than in samples collected prior to cleanout; however, concentrations were still greater than the SQS in three of the four samples. The concentration of BBP was greater than the SQS in one of the grab samples, and the concentration of dimethyl phthalate was greater than the CSL in one sample. The 1,4-dichlorobenzene concentration was greater than the CSL in the first grab sample collected after the pipe was cleaned, but subsequent samples had concentrations below the SQS.

Chemical concentrations in the in-line sediment traps remained fairly consistent from 2003 through 2007 (before and after the cleanout). With the exception of zinc, metals concentrations have been relatively low and generally less than the SQS. Zinc concentrations were greater than the SQS in about half of the sediment trap samples, and greater than the CSL in three samples. BEHP concentrations continue to be above SMS criteria in the trap samples, with concentrations greater than the SQS in about 70% of the in-line trap samples. PAH concentrations in the traps remain low, with HPAH concentrations greater than the SQS in one sample, and LPAH concentrations lower than the SQS in all samples. One of the 45 sediment trap samples collected to date exceeded the SQS criterion for total PCBs.

Storm drain lines and catch basins on the Rainier Commons property[148] (which is within the Diagonal Avenue S CSO/SD basin) were cleaned in 2008 after SPU traced the presence of PCBs in a catch basin to the property in 2004. The source of the PCBs was identified as the exterior building paint that contained approximately 2,300 mg/kg dw PCBs (Vernon Environmental 2006). The buildings were pressure washed and repainted, and the storm drain lines and catch basins on the property were subsequently cleaned. At the same time, SPU cleaned the drain lines on Airport Way S immediately downstream of the property to remove any residual PCB contamination. SPU plans to resample the catch basins in the area after sufficient sediment has accumulated, to confirm whether the source of PCBs has been adequately controlled.

Norfolk CSO/SD Cleanout Program Performance

SPU collected in-line sediment samples in the Norfolk system along Martin Luther King, Jr., Way S before and after the drain lines and catch basins (Map 9-16) were cleaned in the summer of 2005 to remove accumulated sediment in advance of drainage improvements. The initial in-line sediment samples were collected in 2003 and 2005, and were used to evaluate sediment disposal options for the cleaning project. Of the 11 in-line sediment samples, the zinc concentrations were greater than the SQS and the CSL in 8 and 4 samples, respectively. BEHP concentrations were greater than the CSL in 6 of the 14 samples.

RoseWater (2007) collected additional in-line samples in 2007 as part of an investigation to determine the cause of the high rate of sediment deposition observed in the lines. The samples contained lower concentrations of zinc (105 to 603 mg/kg) than did the samples prior to cleanout, but continued to have elevated concentrations of BEHP.

Boeing Plant 2 Storm Drain Cleanout Program

In 2005, Boeing adopted a systematic approach to source control, beginning with an evaluation of the chemical quality of the suspended solids in stormwater discharged

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[148] The Rainier Commons property is occupied by Tully's Coffee.

to the LDW via the Boeing Plant 2 storm water system. Based on sampling of catch basin solids, the evaluation in 2005 identified storm drains with the potential to discharge suspended solids with PCBs and/or SMS metals at concentrations above the CSL. In response to high concentrations of PCBs in two of the storm drains, Boeing removed contaminated solids from these storm drains and blocked further discharge of stormwater to them. The two inactive storm drains (X and Y) were fully decommissioned in 2006.

Between late 2006 and early 2008, Boeing completed two "wet season" rounds of in-line sampling of suspended solids in the five remaining storm drains that were identified as being of interest. The sampling was conducted by pumping and filtering large quantities of stormwater as it passed through downgradient manhole locations prior to discharge. The collection and analyses of the solids was conducted in accordance with EPA-approved work plans. Depending on chemical concentrations in the collected solids, further source control measures would be implemented to control potential sources of contamination. Total PCB concentrations in the solids collected from four of the five storm drains were greater than the 2AET,[149] with an average concentration of 2.8 mg/kg dw. Concentrations of metals were variable, with some CSL exceedances of copper, cadmium, chromium, lead, mercury, and zinc.

During summer 2008, Boeing began cleaning a majority of the Plant 2 storm drain systems that primarily drain pavement areas. As of mid-October 2008, approximately 21,600 LF of pipe and 343 catch basins and manholes had been cleaned, an estimated 80 cy of sediments had been removed, and 231 geotextile inserts had been installed in catch basins to reduce the amount of sediment entering the systems. Weather permitting, storm drain cleaning will likely be completed during November 2008, at which time a total of approximately 27,000 LF of pipe and 400 catch basins and manholes will have been cleaned, and 270 geotextile inserts will have been installed. Round 3 of source control sampling is scheduled to begin in November 2008.

North Boeing Field Storm Drain System Cleanout Information

Regular inspection and cleaning of the storm drain system at North Boeing Field (NBF), including manholes, catch basins, and oil/water separators, is conducted by Boeing, usually on an annual basis. In addition, the storm drain system, including piping, has been cleaned on at least three occasions since the mid-1980s.

As a result of the discovery of PCBs in the Georgetown Steam Plant flume and in portions of the storm drain system at NBF, portions of the storm drain system at NBF near the Georgetown Steam Plant were cleaned in 1985 by Seattle City Light (Seattle City Light 1985). Additional storm drain system cleaning was performed in this area by Boeing in 1986 (The Boeing Company 1986).

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[149] TOC was not analyzed, so comparison with the CSL, which is expressed on an OC-normalized basis, was not possible.

In 1992, approximately 60% of the 7 mi of storm drain system piping, 90 percent of the over 400 manholes, and 81 percent of the over 400 catch basins of the NBF storm drain system were cleaned by Boeing. Problems encountered during the cleaning process, including excess water, blockage problems, and obstructed access, prevented the cleaning of the remainder of the storm drain system (Landau Associates 1993).

Another major storm drain system cleaning event was conducted by Boeing between 2005 and 2008. Storm drain system piping, catch basins, and manholes were pressure washed, an oil/water separator near the Georgetown Steam Plant was decommissioned, and portions of the storm drain system were replaced or were lined with cured-in-place pipe.

Sediment traps installed at seven locations at NBF in 2005 have been sampled periodically by Boeing. Concentrations of PCBs in sediment trap samples collected in 2008 have decreased substantially compared with prior sediment trap samples. In March 2008, total PCB concentrations in the solids collected within these seven sediment traps ranged from 0.42 to 16 mg/kg dw.

Boeing Developmental Center South Storm Drain Cleanout Program

The south storm drain from the Boeing Developmental Center discharges near the Norfolk CSO/SD. PCBs were detected in surface sediments on the cap placed offshore of the Norfolk CSO/SD during post-removal monitoring and raised concerns regarding the potential for recontamination of the clean cap material. Subsequent sampling events and source control evaluations conducted by King County, Ecology, and Boeing indicated that surface sediment in a small area inshore of the Norfolk cap and near the south storm drain outfall had PCB concentrations above SMS criteria and there was concern that erosion of these sediments could potentially contribute to recontamination of the Norfolk cap. Under the MTCA Voluntary Cleanup Program (VCP), Boeing completed sediment sampling to define the extent of PCBs above the SMS criteria and performed sediment removal work in this area (near the south storm drain). As part of this overall project, Boeing conducted source control investigation and elimination efforts within the upland areas of the south storm drain system. Source control for the project has been implemented in sequential phases, beginning with sampling and characterization of upland areas in the southern portion of the Boeing Developmental Center, followed by cleanout of the south storm drain system, installation of a sediment trap/oil-water separator located above the storm drain outfall, and performance monitoring.

Sampling of construction materials in the upland area (including samples of paint, roofing materials, catch basin solids, joint caulking compounds, and other materials in the immediate area) did not identify any materials that could provide an ongoing source of PCBs to the storm drain (Project Performance Corporation 2001).

Given the PCB concentrations detected in selected side-wall samples from the south storm drain, Boeing decided to implement the storm drain cleanout project, which was completed in August 2002 (Project Performance Corporation 2002; as cited in Ecology and Environment 2007a). Approximately 500 ft of 24-in.-diameter concrete pipe was cleaned by jet-rodding. The jet-rod cleaning provides multiple (~15) radial water jets at ~ 2,000 psi to remove sidewall scum from the pipe interior. The quantity of sludge removed was estimated at less than 3 cy. Following completion of the drain system cleaning, a video camera was run through the storm drain line. The video inspection indicated that the pressure-washing technique was effective in removing all of the material from the storm drain pipe, including the side-wall scum accumulated on pipe interior walls, except in one section of the pipe, which was subsequently cleaned at the same time as installation of the stormwater treatment vault. A Vortechs® 9000 unit (stormwater treatment vault) was installed in August 2003 (Landau 2004). The Vortechs® 9000 unit is approved by Ecology as a pretreatment technology for removal of total suspended solids from stormwater runoff (Ecology 2007d). Following installation of the sediment trap, Boeing completed a sediment removal action in the area of the south storm drain outfall in September 2003 (Project Performance Corporation 2003); contaminated sediments were first removed and the area was then capped with clean fill material.

Subsequent to the work noted above, Boeing has completed annual monitoring of both surface sediments on the cap placed just offshore of the south storm drain (2004 to 2009) and solids in the storm drain (2004 to 2008) (CALIBRE 2005, 2006, 2007, 2008, 2009). Total PCB concentrations in surface sediments on the cap have remained consistently low from 2004 through 2009 (all represent samples collected after cleanout of the storm drain) and have generally been below the SQS (16 out of 20 samples were below the SQS, and the maximum was less than 2 times the SQS, over 5 years of sampling); total PCBs were undetected at all three cap stations in the most recent round of sampling. Total PCB concentrations in solids collected from a filter bag deployed in a manhole downstream of the Vortechs® 9000 unit have decreased from a maximum of 12.6 mg/kg dw in 2005 to 2.28 mg/kg dw in 2007 (no sample was collected downstream of the Vortechs® 9000 unit in 2008 because the filter bag had become detached and washed away). The mass loading of total PCBs from this source to the LDW has been estimated to be ≤0.55 g/year in each of the 5 years with data (CALIBRE 2009).

Key Manhole Sampling

King County regularly samples wastewater at key locations in the sewer system that lead to the wastewater treatment plant. Week-long composite sampling is conducted twice per year (during a wet season and a dry season) at three pump stations in the LDW drainage basin: W Marginal Way SW, Duwamish, and E Marginal Way S (near Slip 4). In 2003, King County began analyzing samples for phthalates in addition to metals in wastewater samples. Data are available for samples collected in September 2003 (dry season), April 2004 (wet season), and September 2004 (dry season) for phthalates. Key conclusions for the phthalates were (King County and SPU 2005):

- Concentrations of BEHP (2 to 14 µg/L) in the wet season samples collected from the E Marginal Way S and Duwamish pump stations in April of 2004, were within the range of concentrations typically detected in stormwater samples (1 to 16 µg/L).
- In September 2003, samples collected from the W Marginal Way SW pump station contained a significantly higher BEHP concentration (148 µg/L) than the other two stations (mean ~ 5 µg/L).
- Other phthalates detected in the key manhole samples included BBP (100% of samples), diethylphthalate (97% of samples), di-n-butylphthalate (25% of samples), and dimethylphthalate (1% of samples).

Street Sweeping Pilot Program

In 2006, the City of Seattle implemented a pilot project to evaluate street sweeping as a tool to reduce the amount of pollutants discharged to area receiving water bodies (SPU and Herrera 2009). The pilot test was conducted using regenerative air sweepers at two residential test sites (West Seattle and Southeast Seattle-Columbia City) and one industrial site in the Diagonal/Duwamish drainage basin. Testing began in the residential areas in June 2006 and concluded in June 2007. In the industrial area, testing began in November 2006 and concluded in June 2007. At each site, a 10 to 25 block area was divided into a control area and a test area. In the test area, each side of the street was swept on alternating weeks, but the control area was not swept. Catch basin sediment, sweeper waste, and street dirt accumulations were measured every month from the test and control areas to determine the quantity of material removed by the sweeping effort. Samples of each media type (e.g., catch basin sediment, sweeper waste, and street dirt) were also collected every month and composited each quarter for chemical analysis of metals, SVOCs, PCBs, grain size, and organic carbon.

Test results showed that street sweeping was effective in removing about 2,200 to 3,100 pounds of material per acre of street swept per year. The largest amount of sediment was removed in the Diagonal/Duwamish test site. Similarly, sweeping was effective in reducing the amount of dirt present on the street. Average monthly reductions in street dirt mass were 74, 90, and 48% at the West Seattle, Southeast Seattle, and Diagonal/Duwamish test sites, respectively.

Concentrations of most chemicals (metals, petroleum hydrocarbons, phthalates, and PCBs) were higher in the catch basin samples than in the street dirt and sweeper waste. The highest concentrations were generally detected in the catch basin samples collected from the Diagonal/Duwamish test site. Overall, PCBs were detected in less than 50% of all the samples, with most detections occurring in samples collected from the Diagonal/Duwamish site (SPU and Herrera 2009).

In addition to measuring mass removal rates and quantities, the pilot test was also conducted to evaluate whether street sweeping reduces the rate of sediment accumulation in catch basins, and thereby reduces the frequency that catch basins need to be cleaned. The results indicate that catch basins are relatively ineffective in removing sediment. Over the course of the pilot test, the mass of sediment accumulated in the catch basins accounted for only about 5 to 15% of the total mass removed from the streets. Sweeping did not significantly affect the amount or rate of sediment accumulation in catch basins. The final report (SPU and Herrera 2009) is available on SPU's website.

9.4.5 Spills and leaks

Spills and leaks may enter the LDW through various pathways. This section provides information on data resources for leaks and spills in the LDW drainage basin.

9.4.5.1 Available spill data

Ecology receives spill reports, which are tracked through ERTS, and placed on the Spills Program database. Records of spills in the LDW drainage area currently listed on Ecology's Spills database are listed in Table G-14 in Appendix G. In addition to responding to spills, Ecology regularly refers reports of spills within the LDW area to SPU or the USCG for response.[150] Records of SPU spill responses are included in Table G-15 in Appendix G.

Three significant known historical spills have been reported. One spill occurred at the Duwamish/Diagonal SD. This spill was reportedly an oil spill that originated at a sump at the Union Pacific Railroad. The Duwamish Manufacturing Company also spilled crude oil into the LDW. The site was cleaned but a large portion of the oil had already reached the LDW (WPCC 1945-1957, as cited in Dexter et al. 1985).

In 1974, a spill of high-concentration PCB dielectric fluid occurred in Slip 1 (EPA 1975a). As a result of two separate dredging events, most of the PCBs spilled were recovered, although sampling results indicated that PCBs had spread prior to cleanup (EPA 1975b). Recent sediment samples have detected PCB concentrations in Slip 1 that

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[150] USCG also maintains a database of oil and chemical spills into water bodies that are reported directly to the USCG (i.e., without first being reported to Ecology). The USCG spill database is available through the National Response Center website). The USCG database is organized by year of report and is not geographically coded.

exceed the SQS in surface sediment and the CSL in subsurface sediment (Windward and RETEC 2007).

9.4.5.2 Leaks

Leaks can occur from various sources such as pipes and storage tanks or from industrial or commercial equipment and process operations. Ecology currently regulates and monitors active and inactive LUSTs. The LUST list includes confirmed and suspected leaking tanks owned and operated by entities such as gas stations, manufacturing industries, commercial properties, and government agencies. The LUST list contains sites that have been cleaned up and sites currently in the process. Currently, about 70 of the LUST sites on Ecology's active LUST list are located in the LDW drainage area (EDR 2006b). The records for active and inactive LUST sites are summarized in Tables G-7 and G-8 in Appendix G, respectively.

9.4.5.3 Spill prevention and cleanup programs

Ecology maintains a spill prevention, preparedness, and response program whose goal is to protect public health, safety, and the environment from accidental releases of oil and other hazardous materials (Ecology 2006b). In addition, Ecology provides 24-hr emergency response units statewide and investigates hazardous material incidents in order to identify practices that will reduce the number of future spills. Ecology has the authority to issue fines or require changes in operations at facilities that violate spills regulations.

Ecology partnered with EPA, the USCG, and other state and local agencies from Oregon and Idaho to produce the Northwest Area Contingency Plan. The plan provides guidelines for response actions to spill incidents, and, in Washington State, it functions as the statewide master response plan for oil and hazardous substance releases (EPA and USCG 2006). Regulations calling for a regional response plan capable of responding to hazardous materials spills and other releases are contained in 40 CFR 300.

SPU inspectors routinely respond to water quality complaints throughout the city of Seattle. Complaints are received from a number of sources, including the City's hotline number, staff reports, and referrals from other City departments and agencies. In addition to identifying and controlling the causes of a complaint, City inspectors also provide technical assistance to local residents and businesses regarding BMPs for reducing pollution. SPU also maintains a coordinator network to respond to spills affecting City infrastructure on a 24-hour basis, seven days a week. SPU maintains an on-call contractor to provide emergency cleanup services. Records from the SPU spill coordinator program are provided in Appendix G.

9.4.6 Groundwater

Numerous studies that provide information on groundwater conditions have been conducted in the LDW basin. Groundwater monitoring has been conducted at many facilities adjacent to and nearby the LDW (Map 9-17). Although most of those studies were conducted to assess conditions at specific facilities, they provide data that are useful in evaluating groundwater conditions in the greater LDW basin. Multiple seep and porewater sampling events have also been conducted in the LDW (see Maps 4-11a and 4-12a, respectively). A review of existing groundwater data was conducted for 12 facilities identified by EPA and Ecology in 2003, based on information available at the time, as part of the Phase 1 RI (Groundwater Pathway Assessment, Appendix G [Windward 2003a]) (Map 9-17). This preliminary assessment evaluated the potential for chemicals in groundwater at these 12 facilities to reach sediment in the LDW through groundwater discharge based on data that were available at that time.

Table 9-15 provides a summary of groundwater information for the 12 facilities discussed in Appendix G of the Phase 1 RI as well as numerous other facilities. Additional facilities were selected for inclusion in the table if they were shoreline properties associated with one of the 11 SCAs discussed in Appendix I[151] or if they were identified by Ecology as being sites of interest for groundwater (Hiltner 2008). As part of their source control program, Ecology is also assessing groundwater at upland facilities associated with the SCAs, many of which are not included in Table 9-15. The SCAP and data gaps reports present available groundwater information for all of the facilities associated with the SCAs.

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[151] Of the properties associated with the 11 SCAs discussed in Appendix I, only the shoreline properties were included in Table 9-15. Ecology is evaluating groundwater at several upland facilities associated with these 11 SCAs as part of their source control program. For groundwater information on upland facilities associated with the 11 SCAs that are not included in Table 9-15, see Ecology's SCAPs and data gaps reports.

Table 9-15. Summary of available information for groundwater sites in the LDW

LDW RM	Facility	SCA	Chemicals Above SQS In the RI Baseline Surface Sediment Dataset within the Associated Sediment Area[a]	Chemicals Detected in Groundwater[b]	Chemicals Detected in Seeps	Chemicals Detected in Porewater	Groundwater Program Details and Fate and Transport Information, if Available	Sources of Additional Information
Shoreline Properties								
0.3 – 0.5 E	T-106 (northern portion of parcel, formerly referred to as T-106 W)	Duwamish/ Diagonal Way	cadmium, chromium, lead, mercury, silver, zinc, 2-methylnaphthalene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene, total HPAHs, total LPAHs, BEHP, BBP, dimethyl phthalate, 1,2,4-trichlorobenzene, 1,2-dichlorobenzene, 1,4-dichlorobenzene, 2,4-dimethylphenol, 4-methylphenol, benzoic acid, benzyl alcohol, n-nitrosodiphenylamine, phenol, total PCBs, dioxins and furans[c, d]	arsenic ,lead, toluene, petroleum-related compounds	No seep data were reported.	No porewater data were reported.	Groundwater sampling was conducted as part of an environmental site assessment in 2007; these investigations assessed UST locations and an area of historical cement kiln dust disposal.	Pinnacle Geosciences (2005, 2007)
0.3 – 0.5 E	T-106 (southern portion of parcel, formerly referred to as T-106 SW)	Duwamish/ Diagonal Way	cadmium, chromium, lead, mercury, silver, zinc, 2-methylnaphthalene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene, total HPAHs, total LPAHs, BEHP, BBP, dimethyl phthalate, 1,2,4-trichlorobenzene, 1,2-dichlorobenzene, 1,4-dichlorobenzene, 2,4-dimethylphenol, 4-methylphenol, benzoic acid, benzyl alcohol, n-nitrosodiphenylamine, phenol, total PCBs, dioxins and furans[c, d]	arsenic, lead, petroleum hydrocarbons	No seep data were reported.	No porewater data were reported.	Groundwater sampling was conducted in the 1990s as part of LUST investigations and at a compressor area and a steam cleaning area.	Pinnacle Geosciences (2005)
0.5 – 0.7 E	T-108 (former Chiyoda property)	Duwamish/ Diagonal Way	cadmium, chromium, lead, mercury, silver, zinc, 2-methylnaphthalene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene, total HPAHs, total LPAHs, BEHP, BBP, dimethyl phthalate, 1,2,4-trichlorobenzene, 1,2-dichlorobenzene, 1,4-dichlorobenzene, 2,4-dimethylphenol, 4-methylphenol, benzoic acid, benzyl alcohol, n-nitrosodiphenylamine, phenol, total PCBs, dioxins and furans[c, d]	arsenic, cadmium, chromium, copper, lead, nickel, zinc, 1-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzofluoranthenes, chrysene, fluoranthene, fluorene, naphthalene, phenanthrene, pyrene, total HPAHs, total LPAHs, PCBs, benzene, ethylbenzene, toluene, xylenes	No seep data were reported.	No porewater data were reported.	Numerous soil and groundwater investigations have been conducted at the facility over the past 20 years as part of site assessments and PCB disposal site investigations. A groundwater sampling program was implemented in 2006 and 2007 in accordance with an Ecology approved work plan. Based on the results, it was recommended that additional groundwater monitoring at the facility was not necessary.	Pacific Groundwater Group (2006, 2007), Windward (2003a)
0.7 – 0.9 E	Federal Center South	Duwamish/ Diagonal Way	cadmium, chromium, lead, mercury, silver, zinc, 2-methylnaphthalene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene, total HPAHs, total LPAHs, BEHP, BBP, dimethyl phthalate, 1,2,4-trichlorobenzene, 1,2-dichlorobenzene, 1,4-dichlorobenzene, 2,4-dimethylphenol, 4-methylphenol, benzoic acid, benzyl alcohol, n-nitrosodiphenylamine, phenol, total PCBs, dioxins and furans[c, d]	benzene, ethylbenzene, toluene, xylene, petroleum hydrocarbons	arsenic, cadmium, copper, lead, mercury, nickel, silver, zinc	No porewater data were reported.	Groundwater monitoring in 2000 identified a gasoline plume.	Ecology (2004a), Windward (2004b)
1.4 E	St. Gobain Containers (former St. Gobain Glass)	St. Gobain to Glacier Northwest[e]	zinc, chrysene, benzyl alcohol, phenol, total PCBs[c]	No groundwater data were reported.	No seep data were reported.	No porewater data were reported.	No groundwater monitoring program is in place.	

Table 9-15, cont. Summary of available information for groundwater sites in the LDW

LDW RM	Facility	SCA	Chemicals Above SQS In the RI Baseline Surface Sediment Dataset within the Associated Sediment Area[a]	Chemicals Detected in Groundwater[b]	Chemicals Detected in Seeps	Chemicals Detected in Porewater	Groundwater Program Details and Fate and Transport Information, if Available	Sources of Additional Information
1.5 E	Longview Fibre Paper and Packaging	St. Gobain to Glacier Northwest[e]	zinc, chrysene, benzyl alcohol, phenol, total PCBs[c]	PAHs, SVOCs, petroleum hydrocarbons	No seep data were reported.	No porewater data were reported.	Groundwater monitoring took place in 2003 after remediation of a LUST; details were not available. It is unclear whether the groundwater monitoring is ongoing.	EDR (2006a)
1.6 E	BPB Gypsum/Old James Hardy Gypsum	St. Gobain to Glacier Northwest[e]	zinc, chrysene, benzyl alcohol, phenol, total PCBs[c]	No groundwater data were reported.	arsenic, cadmium, copper, lead, nickel, mercury, zinc, carbon disulfide, TPH (diesel)	No porewater data were reported.	No groundwater monitoring program is in place.	Windward (2004b)
1.75 E (northern parcel) and 1.9 E (southern parcel)	Duwamish Marine Center	Slip 2 to Slip 3[e]	total PCBs[c]	non-halogenated solvents, PCBs, petroleum hydrocarbons	arsenic, cadmium, copper, lead, mercury, nickel, silver, zinc	aluminum, arsenic, barium, cadmium, calcium, iron, lead, magnesium, manganese, potassium, sodium, vanadium, tributyltin as ion	Groundwater sampling conducted as part of site assessment related to CSCSL record; no record of formal groundwater monitoring program identified during assessment.	Cargill and Thomas (2007), EDR (2006b), Weston (1999a)
2.1 E	SCS Refrigerated Services	Slip 3 to Seattle Boiler Works	arsenic, benzyl alcohol[c]	No groundwater data were reported.	No seep data were reported.	No porewater data were reported.	No fate and transport or groundwater program information was available.	
2.1 – 2.3 E	Glacier Marine Services	Slip 3 to Seattle Boiler Works	arsenic, benzyl alcohol[c]	No groundwater data were reported.	No seep data were reported.	aluminum, arsenic, barium, cadmium, calcium, copper, iron, lead, magnesium, manganese, potassium, silver, sodium, vanadium, zinc, dibutyltin as ion, tributyltin as ion	No fate and transport or groundwater program information was available.	Weston (1999a)
2.1 – 2.3 E	Seattle Distribution Center	Slip 3 to Seattle Boiler Works	arsenic, benzyl alcohol[c]	No groundwater data were reported.	No seep data were reported.	No porewater data were reported.	No fate and transport or groundwater program information was available.	
2.2 – 2.25 E	Bunge Foods/Guimont Parcel (Dawn Food Products)	Slip 3 to Seattle Boiler Works, Seattle Boiler Works to Slip 4	arsenic, mercury, fluoranthene, total PCBs, benzyl alcohol[c]	No groundwater data were reported.	1,2-DCE, vinyl chloride[f]	No porewater data were reported.	No fate and transport or groundwater program information was available.	
2.3 E	Seattle Boiler Works	Seattle Boiler Works to Slip 4	mercury, fluoranthene, total PCBs[c]	No groundwater data were reported.	1,1-dichloroethane, 1,2-dichloroethane, 1,2-DCE, cis-1,2-DCE, trans-1,2-DCE, 1,2-dichloropropane, acetone, benzene, chloride, chlorobenzene, PCE, TCE, toluene, vinyl chloride, total xylenes[f]	1,1-dichloroethane, cis-1,2-DCE, trans-1,2-DCE, carbon disulfide, PCE, TCE, vinyl chloride[f]	No groundwater monitoring program is in place.	
2.4 E	Seattle Iron & Metals Corporation	Seattle Boiler Works to Slip 4	mercury, fluoranthene, total PCBs[c]	No groundwater data were reported.	1,1,1-trichloroethane, 1,1-dichloroethane, 1,2-dichloroethene, cis-1,2-DCE, 1,2,4-trimethylbenzene, acetone, PCE, TCE, vinyl chloride[f]	1,2-dichlorobenzene, 1,4-dichlorobenzene, 1,1-DCE, 1,1-dichloroethane, carbon disulfide, chlorobenzene, cis-1,2-dichloroethene, isopropylbenzene, toluene, trans-1,2-dichloroethene, vinyl chloride[f]	No groundwater monitoring program is in place.	
2.6 E	Puget Sound Truck Lines	Seattle Boiler Works to Slip 4	mercury, fluoranthene, total PCBs[c]	No groundwater data were reported.	No seep data were reported.	No porewater data were reported.	No groundwater monitoring program is in place.	
2.7 E	Seattle City Light	Seattle Boiler Works to Slip 4	mercury, fluoranthene, total PCBs[c]	No groundwater data were reported.	No seep data were reported.	No porewater data were reported.	No groundwater monitoring program is in place.	

Table 9-15, cont. Summary of available information for groundwater sites in the LDW

LDW RM	Facility	SCA	Chemicals Above SQS In the RI Baseline Surface Sediment Dataset within the Associated Sediment Area[a]	Chemicals Detected in Groundwater[b]	Chemicals Detected in Seeps	Chemicals Detected in Porewater	Groundwater Program Details and Fate and Transport Information, if Available	Sources of Additional Information
2.8 E	Crowley Marine Services	Seattle Boiler Works to Slip 4, Slip 4	mercury, acenaphthene, benzo(a)anthracene, benzo(a)pyrene, benzofluoranthenes, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, total HPAHs, BBP, BEHP, total PCBs[c]	arsenic, copper, 2-methylnaphthalene, acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzofluoranthenes, chrysene, fluoranthene, fluorene, naphthalene, phenanthrene, pyrene BEHP, cis-1,2-DCE, acetone, dichloromethane	total PCBs	arsenic, barium, cadmium, lead, vanadium[g]	Groundwater sampling conducted in the 1980s and 1990s as part of site assessment; samples were collected from 12 wells and analyzed for metals, VOCs, SVOCs, TPH, chlorinated phenols, pesticides, and PCBs. There are plans for an additional groundwater investigation to be conducted on the facility.	SAIC (2006, 2007a, 2008a)
2.95 E	First South Properties	Slip 4	mercury, acenaphthene, benzo(a)anthracene, benzo(a)pyrene, benzofluoranthenes, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, total HPAHs, BBP, BEHP, total PCBs[c]	arsenic, copper, zinc, TPH, 2-methylnaphthalene, acenaphthene, fluorene, phenanthrene, pyrene	mercury, copper	No porewater data were reported.	Groundwater sampling conducted in the 1980s and 1990s as part of site assessment and UST removals; samples were collected from 7 wells and analyzed for metals, VOCs, SVOCs, TPH, and PCBs. The site was granted an NFA in 1997 and groundwater monitoring was terminated in 1998. No record of an ongoing groundwater monitoring program was identified during this assessment.	Ecology (2006a), SAIC (2007a), Windward (2004b)
2.9 – 3.6 E	Boeing Plant 2	Slip 4, Boeing Plant 2/ Jorgensen Forge	cadmium, chromium, copper, lead, mercury, silver, zinc, acenaphthene, benzo(a)anthracene, benzo(a)pyrene, benzofluoranthenes, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, dibenzofuran, fluoranthene, fluorene, Indeno(1,2,3-cd)pyrene, phenanthrene, total HPAHs, total LPAHs, BBP, BEHP, phenol, total PCBs, dioxins and furans[c, d]	aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, molybdenum, nickel, selenium, silver, thallium, vanadium, zinc, naphthalene, BEHP, di-n-buytlphthalate, 2,4-dimethylphenol, 2-methylphenol, 4-chloro-3-methylphenol, 4-methylphenol, phenol, methyl isobutyl ketone, total PCBs, 1,1,1-trichloroethane, 1,1,2-trichloroethane, 1,1,2-trichloro-trifluoroethane, 1,1-DCE,1,1-dichloroethane, 1,2-dichloroethane, cis-1,2-DCE, trans-1,2-DCE, 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, 4-isopropyltoluene,acetone, benzene, bromodichloromethane, carbon disulfide, carbon tetrachloride, chlorobenzene, chloroethane, chloroform, chloromethane, ethylbenzene, isopropylbenzene, n-butylbenzene, n-propylbenzene, PCE, sec-butylbenzene, TCE, tert-butylbenzene, toluene, trichlorofluoromethane, vinyl chloride, m,p-xylene, o-xylene, TPH (diesel, gasoline, and motor oil), 2-butanone, cyanide, methylene chloride	arsenic, chromium, copper, lead, mercury, selenium, zinc, fluoranthene, phenanthrene, total PCBs, cis-1,2-DCE, benzene, TCE, vinyl chloride	1,1-dichloroethene, cis-1,2-DCE, TCE, vinyl chloride[h]	Weston's (1998) mass loading analysis indicated no predicted accumulation in sediments for metals following bank removal. V PCBs were detected in seeps. The likely source of PCBs in seeps is sediment. VOC concentrations detected in porewater samples were less than toxicity thresholds for benthic invertebrates. Groundwater monitoring is on-going.	Ecology (2006a), Ecology and Environment (2007b), Environmental Partners (2008), SAIC (2007b), Weston (1998), Windward (2003a, 2006a)
3.6 – 3.7 E	Jorgensen Forge	Boeing Plant 2/Jorgensen Forge	cadmium, chromium, copper, lead, mercury, silver, zinc, acenaphthene, benzo(a)anthracene, benzofluoranthenes, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, dibenzofuran, fluoranthene, fluorene, Indeno(1,2,3-cd)pyrene, phenanthrene, total HPAHs, total LPAHs, BBP, BEHP, phenol, total PCBs, dioxins and furans[c, d]	cadmium, benzene, ethylbenzene, toluene, vinyl chloride, xylenes, TPH (diesel)	arsenic, cadmium, copper, lead, mercury, nickel, silver, zinc	1,1-dichloroethene, cis-1,2-DCE, TCE, vinyl chloride[h]	No formal groundwater monitoring program is in place. Groundwater sampling was conducted as part of site investigations and/or tank removal.	Ecology and Environment (2007b), Environmental Partners (2008), Windward (2004b)
3.7-3.8 E	Boeing Isaacson	Boeing Isaacson/ Central KCIA	arsenic, acenaphthene, benzo(a)anthracene, benzo(a)pyrene, benzofluoranthenes, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, dibenzofuran, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, phenanthrene, total HPAHs, total LPAHs, BEHP, BBP, benzoic acid total PCBs[c]	arsenic, cadmium, chromium, copper, lead, zinc	arsenic, cadmium, copper, lead, mercury, nickel, silver and zinc[i, j]	No porewater data were reported.	Several groundwater investigations have been conducted, including a hydrogeologic investigation in 2000. Apparent natural attenuation was noted based on a comparison of data from well pairs. Groundwater sampling was conducted in September 2007 and future sampling is planned.	Ecology (2008a), SAIC (2008b), Windward (2003a)
3.8 E	Boeing Thompson	Boeing Isaacson/ Central KCIA	arsenic, acenaphthene, benzo(a)anthracene, benzo(a)pyrene, benzofluoranthenes, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, dibenzofuran, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, phenanthrene, total HPAHs, total LPAHs, BEHP, BBP, benzoic acid, total PCBs[c]	arsenic	arsenic[j]	No porewater data were reported.	Groundwater sampling was conducted between 1988 and 2007. Additional groundwater sampling is planned.	Ecology (2008a), SAIC (2008b), Windward (2003a)

Table 9-15, cont. Summary of available information for groundwater sites in the LDW

LDW RM	Facility	SCA	Chemicals Above SQS In the RI Baseline Surface Sediment Dataset within the Associated Sediment Area[a]	Chemicals Detected in Groundwater[b]	Chemicals Detected in Seeps	Chemicals Detected in Porewater	Groundwater Program Details and Fate and Transport Information, if Available	Sources of Additional Information
3.9 – 4.0 E	PACCAR (former Kenworth Truck Co.)	Slip 6	lead, BBP, total PCBs[k]	arsenic, barium, chromium, copper, nickel, selenium, zinc, benzo(g,h,i,)perylene, chrysene, BEHP, benzyl alcohol, fluoranthene, total PCBs, 1,1,1-trichloroethane, 1,1,2-trichloroethane, 1,1-DCE,1,1-dichloroethane, 1,2-dichloroethane, cis-1,2-DCE, trans-1,2-DCE, acetone, benzene, chloroethane, chloroform, dichloromethane, methylene chloride, PCE, TCE, vinyl chloride, TPH (diesel, gasoline, and motor oil), glycols	arsenic, copper, lead	No porewater data were reported.	VOCs and TPH are likely attenuated, and arsenic is in the range of background concentrations. Groundwater monitoring for VOCs occurred between 1986 and 2002. Two rounds of groundwater monitoring were completed in 2006.	Ecology and Environment (2008b), Windward (2003a)
4.0 -4.2 E	Former Rhône-Poulenc facility	Slip 6	mercury, acenaphthene, benzo(g,h,i)perylene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, total HPAHs, BEHP, benzoic acid, phenol, total PCBs[k]	arsenic, beryllium, cadmium, chromium, copper, lead, mercury, selenium, silver, thallium, vanadium, zinc, benzo(a)pyrene, chrysene, BEHP, 2-methylphenol, 4-methylphenol, bis(2-chloroethyl)ether, PCP, 1,1-dichloroethane, acetone, benzene, dichloromethane, ethylbenzene, formaldehyde, toluene, xylenes	aluminum, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, lead, magnesium, mercury, nickel, vanadium, zinc, BEHP, formaldehyde	antimony, arsenic, chromium, barium, cadmium, copper, iron, lead, magnesium, manganese, mercury, vanadium, zinc, tributyltin, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, fluoranthene, pyrene, BEHP, diethylphthalate, 1,1-biphenyl, caprolactam	Groundwater investigations at the site began in 1985. As of 2007, over 10 years of quarterly groundwater monitoring has been conducted on the site. Toluene contamination in groundwater is still being investigated by EPA.	Ecology and Environment (2008b), EPA (2005d), Rhône-Poulenc (1996), Weston (1999a), Windward (2003a)
4.2 – 4.8 E	Boeing Developmental Center	Slip 6, Norfolk CSO/SD	lead, acenaphthene, total PCBs[k]	aluminum, arsenic, barium, calcium, cobalt, copper, iron, lead, magnesium, molybdenum, nickel, zinc, naphthalene, 1,1,1-trichloroethane, 1,1,2-trichloro-1,2,2-trifluoroethane, 1,1-dichloroethane, 1,2,4-trimethylbenzene, cis-1,2-DCE,1,3,5-trimethylbenzene, 2-hexanone, benzene, bromomethane, cis-1,3-dichloropropene, ethylbenzene, m,p-xylene, o-xylene, PCE, toluene, TCE, vinyl chloride, TPH (diesel, gasoline)	No seep data were reported.	No porewater data were reported.	Plumes of chlorinated solvents are reported to be locally contained and to not extend to the LDW. Natural attenuation expected for hydrocarbons and benzene (500 ft from LDW). Groundwater monitoring occurred as part of an Ecology NFA finding in 2001. Groundwater monitoring was discontinued in 2004.	Ecology and Environment (2007a, 2008b), Windward (2003a)
4.6 E	Boeing Military Flight Center	Norfolk CSO/SD	fluoranthene, BBP, BEHP, 1,4-dichlorobenzene, total PCBs[c]	No groundwater data were reported.	No seep data were reported.	No porewater data were reported.	No groundwater monitoring program is in place.	
1.28 W	Alaskan Marine Lines	Glacier Bay	arsenic, copper, lead, mercury, zinc, acenaphthene, benzo (g,h,i) pyrene, chrysene, fluoranthene, indeno (1,2,3-cd) pyrene, phenanthrene, total HPAHs, BBP, BEHP, PCP, total PCBs, dioxin and furans[c, d]	1-methylnaphthylene, 2-methylnaphthalene, acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, phenanthrene, pyrene, benzene, xylenes (total), TPH (diesel, gasoline, motor oil)	No seep data were reported.	No porewater data were reported.	Groundwater sampling has been conducted as part of site assessment and UST investigations; no formal groundwater monitoring program is in place.	SAIC (2007d), Ecology (2007i)
1.35 W	Duwamish Shipyard	Glacier Bay	arsenic, copper, lead, mercury, zinc, acenaphthene, benzo (g,h,i) pyrene, chrysene, fluoranthene, indeno (1,2,3-cd) pyrene, phenanthrene, total HPAHs, BBP, BEHP, PCP, total PCBs, dioxin and furans[c, d]	arsenic, chromium, lead, benzo(a)pyrene, benzene, vinyl chloride	No seep data were reported.	No porewater data were reported.	No formal groundwater monitoring program is in place. Groundwater information collected from borings not completed as monitoring wells.	SAIC (2007d), Ecology (2007i)
1.45 W	Glacier NW (former Reichhold Chemical)	Glacier Bay	arsenic, copper, lead, mercury, zinc, acenaphthene, benzo-(g,h,i) -pyrene, chrysene, fluoranthene, indeno (1,2,3-cd)-pyrene, phenanthrene, total HPAHs, BBP, BEHP, PCP, total PCBs, dioxin and furans[c, d]	arsenic, silver, naphthalene, 2,4,6-trichlorophenol, 2,4-dicholorpohenol, 2-chloropheonol, PCP	arsenic, copper, lead, mercury, silver, and zinc	No porewater data were reported.	Three wells installed in perched groundwater zone near former Reichhold waste pits; exact locations unknown. No formal groundwater monitoring program is in place.	SAIC (2007d), Ecology (2007i)
1.5 W	MRI Corporation	Glacier Bay	arsenic, copper, lead, mercury, zinc, acenaphthene, benzo (g,h,i) pyrene, chrysene, fluoranthene, indeno (1,2,3-cd) pyrene, phenanthrene, total HPAHs, BBP, BEHP, PCP, total PCBs, dioxin and furans[c, d]	No groundwater data were reported.	No seep data were reported.	No porewater data were reported.	No groundwater monitoring program is in place; however, the Port of Seattle is entering this site into Ecology's VCP program, and a new site investigation including groundwater will begin soon.	SAIC (2007d), Ecology (2007i)
2.2 W	Douglas Management Company	Trotsky Inlet	lead, mercury, zinc, BBP, BEHP, total PCBs, dioxins and furans[c, d]	No groundwater data were reported.	No seep data were reported.	No porewater data were reported.	No groundwater program is in place.	Ecology (2007h), SAIC (2007b, c), Windward (2004b)

Table 9-15, cont. **Summary of available information for groundwater sites in the LDW**

LDW RM	Facility	SCA	Chemicals Above SQS In the RI Baseline Surface Sediment Dataset within the Associated Sediment Area[a]	Chemicals Detected in Groundwater[b]	Chemicals Detected in Seeps	Chemicals Detected in Porewater	Groundwater Program Details and Fate and Transport Information, if Available	Sources of Additional Information
2.2 W	Trotsky (former Northwest Cooperage, present-day Industrial Container Services)	Trotsky Inlet	lead, mercury, zinc, BBP, BEHP, total PCBs, dioxins and furans[c, d]	antimony, arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, zinc, 2-methylnapthalene, acenaphthene, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzofuran, fluoranthene, fluorene, ideno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene, BBP, BEHP, di-n-octyl phthalate,1,2,4-trichlorobenzene, 1,2-dichlorobenzene, 2,4-dimethylphenol, 2-methylphenol, 4-chloro-m-cresol, 4-methylphenol, benzoic acid, hexachlorobenzene, n-nitrosodiphenylamine, PCP, phenol, phenols (total), methyl isobutyl ketone, total PCBs, 2,4′-DDE, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, cis-chlordane, gamma-chlordane, lindane, 1,1-dichloroehtane, trans-1,2-dichloroethylene, acetone, benzene, dichloromethane, ethylbenzene, toluene, vinyl chloride, xylenes, TPH (diesel and gasoline), cyanide	arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, zinc, benzo(a)anthracene, fluoranthene, naphthalene, pyrene, diethylphthalate, dimethylphthalate, di-N-buytlphthalate, pentachlorophenol, 1,2-dichlorobenze, 1,3-dichlorobenzene, 1,4-dichlorobenzene, total PCBs, 2,4′-DDD, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, lindane, carbon disulfide, chlorobenzene, TPH (diesel)	No porewater data were reported.	Groundwater samples were collected as part of Phase I, II, and III soil and groundwater assessments; samples were collected from monitoring wells in 2007.	Ecology (2007h), SAIC (2007b, c)
2.3 W	Boyer Towing properties (three parcels)	Trotsky Inlet	lead, mercury, zinc, BBP, BEHP, total PCBs, dioxins and furans[c, d]	No groundwater data were reported.	No seep data were reported.	No porewater data were reported.	No groundwater program is in place.	
3.0 – 3.1 W (inland and shoreline portions)	Long Painting	none[l]	PCBs, hexachlorobenzene[k]	arsenic, barium, chromium, lead, 1,1,1-trichloroethane, PCE	arsenic, cadmium, copper, lead, mercury, nickel, silver, zinc	tributyltin	Groundwater sampling has been conducted at the facility; however, no information was available regarding ongoing or future groundwater monitoring. No plume was identified on site, but groundwater data were insufficient. Metal groundwater COCs have not accumulated to concentrations above the SQS in adjacent sediments. The two VOC COCs were not detected in the single sediment sample analyzed for VOCs.	SEA (1998), Windward (2003a, 2004b)
3.4 – 3.5 W	South Park Marina	Terminal- 117	2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, dibenzofuran, fluoranthene, fluorene, indeno (1,2,3-cd)pyrene, phenanthrene, total HPAHs, total LPAHs, benzyl alcohol, phenol, total PCBs[c]	No groundwater data were reported.	No seep data were reported.	No porewater data were reported.	No groundwater program is in place.	
3.5 – 3.7 W	T-117	Terminal- 117	2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, dibenzofuran, fluoranthene, fluorene, indeno (1,2,3-cd)pyrene, phenanthrene, total HPAHs, total LPAHs, benzyl alcohol, phenol, total PCBs[c]	arsenic, cadmium, chromium, copper, lead, silver, zinc, 1-methylnaphthalene, acenaphthene, benzo(a)anthracene, total Benzofluoranthenes, chrysene, fluorene, naphthalene, total HPAHs, total LPAHs, BEHP, total PCBs, cis-1,2-dichloroethene, acetone, chlorobenzene, tetrachloroethene, trichloroethene, o-xylene, m,p-xylene, total xylenes, TPH (diesel, lube oil, and motor oil)	chromium, copper, zinc, BEHP, total PCBs	No porewater data were reported.	Quarterly sampling was conducted in 1997-1998. Several groundwater investigations have been conducted from 1991 to 2006. Groundwater monitoring is ongoing.	Windward (2003a) Windward et al. (2005a; 2008)
3.67 – 3.91 W	Boeing South Park	Terminal- 117	2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, dibenzofuran, fluoranthene, fluorene, indeno (1,2,3-cd)pyrene, phenanthrene, total HPAHs, total LPAHs, benzyl alcohol, phenol, total PCBs[c]	No groundwater data were reported.	No seep data were reported.	No porewater data were reported.	No groundwater program is in place.	

Table 9-15, cont. Summary of available information for groundwater sites in the LDW

LDW RM	FACILITY	SCA	CHEMICALS ABOVE SQS IN THE RI BASELINE SURFACE SEDIMENT DATASET WITHIN THE ASSOCIATED SEDIMENT AREA[a]	CHEMICALS DETECTED IN GROUNDWATER[b]	CHEMICALS DETECTED IN SEEPS	CHEMICALS DETECTED IN POREWATER	GROUNDWATER PROGRAM DETAILS AND FATE AND TRANSPORT INFORMATION, IF AVAILABLE	SOURCES OF ADDITIONAL INFORMATION
Inland Properties								
0.7 – 0.9 E	Union Pacific Railroad Argo fueling site	Duwamish/ Diagonal Way	cadmium, chromium, lead, mercury, silver, zinc, 2-methylnaphthalene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene, total HPAHs, total LPAHs, BEHP, BBP, dimethyl phthalate, 1,2,4-trichlorobenzene, 1,2-dichlorobenzene, 1,4-dichlorobenzene, 2,4-dimethylphenol, 4-methylphenol, benzoic acid, benzyl alcohol, n-nitrosodiphenylamine, phenol, total PCBs, dioxins and furans[c, d]	TPH (diesel)	No seep data were reported.	No porewater data were reported.	Groundwater monitoring wells were installed to monitor a diesel plume present beneath the facility. The plume migrates to the west. Groundwater remediation operations, including pumping and air sparging, were conducted between 2001 and 2004; over 38,000 gal. of diesel were recovered.	Ecology (2004a), SAIC (2007j)
1.4 E	Philip Services Corp./ Burlington Environmental	none[l]	mercury, zinc, chrysene, total PCBs, benzyl alcohol, phenol[k]	barium, cadmium, chromium, copper, lead, manganese, nickel, selenium, silver, vanadium, benzo(a)anthracene, benzofluoranthenes (total), benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, indeno(1,2,3-cd)pyrene, benzoic acid, 1,1-DCE, 1,1-dichloroethane, 1,1,1-trichloroethane, 1,2-DCE, cis-1,2-DCE, trans-1,2-DCE, 1,2,4-trimethylbenzene, benzene, chlorobenzene, chloroethane, dichloromethane, ethylbenzene, PCE,TCE, toluene, vinyl chloride, xylenes, petroleum hydrocarbons, 2-butanone, ethane, methane	arsenic, cadmium, copper, lead, mercury, nickel, silver, zinc, TPH-diesel range, TPH	aluminum, arsenic, barium, lead, manganese, vanadium, tributyltin	Site-wide FS completed in 2006 concluded that chlorinated ethenes, 1,4-dioxane and possibly vinyl chloride, could reach the LDW at concentrations greater than the surface water protection criteria.[m] As of 2002, quarterly groundwater monitoring was ongoing. In 2004, a subsurface barrier wall and groundwater recovery system were installed at the facility. As of 2006, this system was still being monitored.	PSC (2001a, b, 2002a, b, c), Geomatrix (2006)
2.35 E	Great Western International (Fox Avenue Buildings)	Seattle Boiler Works to Slip 4	total PCBs[k]	aluminum, arsenic, barium, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, zinc, 2-chloronaphthalene, 2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, chrysene, dibenzofuran, fluoranthene, fluorene, naphthalene, phenanthrene, pyrene, BEHP, diethylphthalate, dimethyl phthalate, di-n-butyl phthalate, 1,2-dichlorobenzene, 1,3-dichlorobenzene, 1,4-dichlorobenzene, 2,3,5-trichlorophenol, 2,4,5-trichlorophenol, 2,4-dichlorophenol, 2,4-dimethylphenol, 2-methylphenol, 4,6-dinitro-2-methylphenol, 4-chloro-3-methylphenol, 4-methylphenol, benzoic acid, benzyl acid, carbazole, isophorone, n-nitro-di-n-propylamine, n-nitroso-di-n-propylamine, PCP, phenol, 1,1,1,2-trichloroethane, 1,1,1-trichloroethane, 1,1,2-trichloroethane, 1,1-dichloroethane, 1,2-dichloroethane, 1,1-DCE, 1,2-DCE, cis-1,2-dichloroethane, cis-1,2-DCE, trans-1,2-DCE, 1,2,4-trimethylbenzene, 1,2-dichloropropane, 1,3,5-trimethylbenzene, acetone, benzene, carbon disulfide, chloroform, DCE, ethane, ethene, ethylbenzene, iso-propyl benzene, iso-propyl toluene, n-butyl benzene, n-propyl benzene, PCE, sec-butyl benzene, styrene, TCE, tetrachlorophenols (total), toluene, trichlorofluoromethane, vinyl chloride, m,p- xylene, total xylenes, TPH (diesel and gasoline), 1,2-propanediol, 2-butanone, 2-propanol, 4-methyl-2-pentanone, diethylene glycol, ethylene glycol, methane, methanol, methylene chloride, sulfate	acenaphthene, total LPAH, total PAH, 1,1,1-trichloroethane, 1,1-DCE, 1,1-dichloroethane, 1,2,4-trimethylbenzene, 1,2-dichlorobenzene, 1,2-DCE (total), cis-1,2-DCE, trans-1,2-DCE, 1,2-dichloroethane, 1,2-dichloropropane, acetone, benzene, chlorobenzene, PCE, toluene, total xylenes, TCE, vinyl chloride, o-xylene	1,2-dichlorobenzene, 1,4-dichlorobenzene, 1,1-DCE, 1,1-dichloroethane, 1,2-dichloroethane, cis-1,2-DCE, trans-1,2-DCE, 1,2-dichloropropane, benzene, carbon disulfide, chlorobenzene, isopropylbenzene, PCE, TCE, toluene, vinyl chloride	Fate and transport analyses for VOCs indicated extensive degradation prior to discharge. In 2004, an underground injection control cleanup program was implemented at the site. As of 2006, the cleanup was still being conducted and its effectiveness was under evaluation. It was not specified whether these evaluations include a groundwater monitoring program. All VOC concentrations in porewater were less than toxicity thresholds for benthic invertebrates except for two cis1,2-DCE concentrations in the high intertidal, which were greater than the no-effect threshold but less than the effect threshold.	SAIC (2008a), Terra Vac and Floyd & Snider (2000), Windward (2003a), Windward (2006a)
2.7 E	Georgetown Steam Plant	Slip 4	mercury, acenaphthene, benzo(a)anthracene, benzo(a)pyrene, benzofluoranthenes, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, total HPAHs, BBP, BEHP, total PCBs[c]	PCBs, trichloroethylene	No seep data were reported.	No porewater data were reported.	Five groundwater monitoring wells were installed and quarterly groundwater monitoring (to continue for one year) was initiated in August 2006.	Ecology (2007j), SAIC (2007a)
2.7 – 3.35 E	North Boeing Field	Slip 4	mercury, acenaphthene, benzo(a)anthracene, benzo(a)pyrene, benzofluoranthenes, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, total HPAHs, BBP, BEHP, total PCBs[c]	antimony, arsenic, beryllium, chromium, copper, lead, nickel, zinc, BEHP, total PCBs, cis-1,2-DCE, acetone, benzene, dichloromethane, ethylbenzene, PCE, TCE, toluene, vinyl chloride, total xylenes, TPH	No seep data were reported.	No porewater data were reported.	Groundwater monitoring has been conducted at several locations on NBF as part of previous site investigations. Ongoing semi-annual groundwater monitoring is occurring at two VOC locations and two TPH locations: nine wells are monitored for VOCs (cis-1,2-DCE, PCE, TCE, vinyl chloride), and six wells are monitored for TPH and BTEX.	SAIC (2007e)

Table 9-15, cont. Summary of available information for groundwater sites in the LDW

LDW RM	Facility	SCA	Chemicals Above SQS In the RI Baseline Surface Sediment Dataset within the Associated Sediment Area[a]	Chemicals Detected in Groundwater[b]	Chemicals Detected in Seeps	Chemicals Detected in Porewater	Groundwater Program Details and Fate and Transport Information, if Available	Sources of Additional Information
2.6 W	South Park Landfill	none[l]	acenapthene, anthracene, benzo(a)anthracene, chrysene, dibenzofuran, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, phenanthrene, total HPAHs, total LPAHs, BBP, BEHP, benzyl alcohol, benzoic acid, PCBs, hexachlorobenzene[k]	aluminum, arsenic, barium, calcium, copper, iron, magnesium, manganese, potassium, selenium, cis-1,2-DCE, chlorobenzene, and vinyl chloride	arsenic, cadmium, copper, lead, mercury, nickel, silver, zinc	No porewater data were reported.	Vinyl chloride not expected to discharge at concentrations of concern because of relatively low concentrations and distance to LDW. Several rounds of groundwater monitoring have been conducted since 1997 and additional monitoring will likely be conducted in the future as part of a planned RI/FS at the site.	Windward (2003a, 2004b)
3.6 W	Basin Oil	Terminal- 117	2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, dibenzofuran, fluoranthene, fluorene, indeno (1,2,3-cd)pyrene, phenanthrene, total HPAHs, total LPAHs, benzyl alcohol, phenol, total PCBs[c]	aluminum, antimony, barium, beryllium, chromium, cobalt, copper, iron, lead, magnesium, manganese, mercury, nickel, selenium, silver, thallium, vanadium, zinc, PCBs, 1,1,1-trichloroethane, 1,1,2-trichloroethane, 1,1-DCE, 1,1-dichloroethane, cis-1,2-DCE, trans-1,2-DCE, acetone, chloroform, chloromethane, chloroform, DCE, PCE, TCE, cyanide	See the seep information provided for T-117. Basin Oil is located adjacent to T-117; groundwater from the two facilities likely discharges to the LDW in approximately the same zone.	No porewater data were reported.	Quarterly sampling was conducted in 1997-1998. Two single groundwater sampling events were conducted in 1991 and 1994.	Windward et al. (2003)
4.1 W	Advance Electroplating	none[l]	no exceedances within the zone of potential groundwater discharge[k]	aluminum, antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, selenium, silver, thallium, vanadium, zinc, acetone, chloroform, chloromethane, PCE, TCE, (and their daughter products), cyanide	No seep data were reported.	arsenic, barium, copper, lead, manganese, vanadium	The site is a significant distance from LDW but may discharge to Hamm Creek first. Biodegradation and volatilization of VOCs is expected between the site and the LDW. No information was identified regarding an ongoing groundwater monitoring program; the most recent groundwater monitoring data are from February 1999.	Windward (2003a)

Note: Information was provided in the groundwater pathway assessment (Appendix G in Windward 2003a), the seep (Windward 2004b) and porewater (Windward 2006a) studies conducted for the RI, additional seep and porewater studies conducted as part of other investigations, documents prepared by Ecology as part of their LDW source control program, including SCAPs and data gap reports, and in additional source documents provided by LDWG members.

[a] Most of the chemical lists are based on SQS exceedances of RI baseline surface sediment data within the administrative SCA surface sediment boundaries established by Ecology in February 2007; however, some are based on SQS exceedances within the zones of potential groundwater discharge estimated for facilities in the groundwater pathway assessment (Appendix G in Windward 2003a). Footnotes c and k specify which lists were generated using the SCA administrative boundaries and which were generated using the zones of potential groundwater discharge.

[b] Includes chemicals detected in historical samples and more recent samples.

[c] The list of chemicals is based on SQS exceedances of RI baseline surface sediment data within the administrative SCA surface sediment boundaries established by Ecology in February 2007. Chemical lists for facilities associated with more than one SCA include all chemicals detected above the SQS in both SCAs. The lists of chemicals detected above the SQS in surface sediment for T-108, Boeing Plant 2, Boeing Isaacson, and T-117 are based on exceedances within the administrative SCA surface sediment boundaries established by Ecology in February 2007, even though zones of potential groundwater discharge were established for these facilities in the groundwater pathway assessment (Appendix G in Windward 2003a). The SCA boundaries were used because they covered approximately the same area as the zones of potential groundwater discharge (Appendix G in Windward 2003a).

[d] SMS criteria do not exist for dioxins and furans. Dioxins and furans were included because they are a risk driver chemical with highly elevated concentrations (i.e., TEQ > 100 ng/kg dw) in surface sediment in the associated SCA.

[e] The St. Gobain to Glacier Northwest and Slip 2 to Slip 3 SCAs were two of the 23 SCAs identified by Ecology, but they are not included among the 11 SCAs discussed in Appendix I. The rationale for the selection of the 11 SCAs is discussed in Appendix I. Facilities within these two SCAs are included in the table because they were identified by Ecology as facilities of interest for groundwater.

[f] The seep and porewater samples collected near the Bunge Foods/Guimont Parcel (Dawn Food Products) shoreline, the Seattle Iron and Metals shoreline, and the Seattle Boiler Works shoreline (see seep sampling locations between RM 2.2 and RM 2.5 E on Maps 4-11a and 9-17 and porewater sampling locations between RM 2.3 and RM 2.45 E on Maps 4-12a and 9-17) were collected as part of investigations to characterize the Great Western International facility. These samples are also included in the information for the Great Western International (Fox Avenue Buildings) facility.

[g] Porewater sample was collected near the middle of Slip 4 (sample DR181 on Map 4-12a).

[h] Porewater samples were collected near the Boeing Plant 2 and Jorgensen Forge property boundary (see sampling locations near RM 3.6 E on Map 4-12a and Map 9-17). Data from these samples are presented for both facilities.

[i] The seep sample in which arsenic, cadmium, copper, lead, mercury, nickel, silver, and zinc were detected was collected along the Jorgensen Forge shoreline (sample SP-20 on Map 4-11a) near the property boundary between Jorgensen Forge and Boeing Isaacson. Data for this seep sample are presented for both facilities.

[j] One seep sample was collected near the Boeing Isaacson and Boeing Thompson property boundary (sample Seep-1 on Map 4-11a). Data from this sample are presented in the table for both facilities.

[k] The lists of chemicals detected above the SQS in surface sediment for these facilities are based on exceedances within the zone of potential groundwater discharge estimated for each facility in the groundwater pathway assessment (Appendix G in Windward 2003a).

[l] Facility has not yet been associated with an SCA by Ecology.

[m] The final RI cleanup levels selected for Philip Services Corp. for the water table, shallow, and intermediate groundwater were the minimum concentrations based on the following: calculated MTCA Method B groundwater cleanup levels based on an Asian and Pacific Islander exposure scenario for the consumption of fish for the groundwater-to-surface water exposure pathway; AWQC based only on the human health consumption of organisms (Section 304 of the federal CWA); ERA surface water screening levels protective of aquatic biota in surface water; AWQC freshwater and marine criteria maximum concentration, criteria continuous concentration, and organoleptic effects (Section 304 of the federal CWA); Washington State freshwater and marine acute and chronic effects criteria; and MTCA Method A cleanup levels (Geomatrix 2006).

Shading identifies the 12 facilities specifically discussed in Section 9.4.6.

AWQC – ambient water quality criteria
BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
COC – chemical of concern
CSCSL – Confirmed and Suspected Contaminated Sites List
CSO – combined sewer overflow
DCE – dichloroethene
DDD– dichlorodiphenyldichloroethane
DDE –dichlorodiphenyldichloroethylene
DDT– dichlorodiphenyltrichloroethane
dw – dry weight
FS – feasibility study
GWI – Great Western International
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LDW – Lower Duwamish Waterway
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LUST – leaking underground storage tank
NBF – North Boeing Field
NFA – no further action
PCB – polychlorinated biphenyl
PCE – tetrachloroethene
PCP –pentachlorophenol
PSC – Phillip Services Corporation
RI – remedial investigation
RM – river mile
SAIC – Science Applications International Corporation
SCA – source control area
SCAP – source control action plan
SD – storm drain
SQS – sediment quality standards
SRI – supplemental remedial investigation
SVOC – semivolatile organic compound
T-108 – Terminal 108
T-117 – Terminal 117
TCE – trichloroethene
TPH – total petroleum hydrocarbon
UST – underground storage tank
VOC – volatile organic compound

Information in Table 9-15 was provided in the seep (Windward 2004b) and porewater (Windward 2006a) studies performed as part of the RI,[152] additional seep and porewater studies conducted as part of other investigations, documents prepared by Ecology as part of their LDW source control program, or in additional source documents provided by LDWG. Table 9-15 lists chemicals detected in groundwater, seeps, and porewater, when available, for each of the facilities. Table 9-15 also includes a list of chemicals that were detected above the SQS in surface sediment samples in the baseline dataset within the administrative boundary of the associated SCA or within the zone of potential groundwater discharge for the 12 facilities evaluated as part of the groundwater pathway assessment (Appendix G in Windward 2003a).[153] Fate and transport and groundwater program information is also provided when available. Key references are listed for additional information. The information presented in Table 9-15 and in this section is not intended to be a complete assessment of the chemical contamination in groundwater at the listed facilities or the potential for chemicals in groundwater to present a risk to sediment-dwelling organisms. It is intended to be a summary of information available at the time this report was drafted on chemicals present in groundwater.

It is important to note that additional reference documents containing groundwater data and other information exist for many of the facilities listed in Table 9-15; therefore, the summary may not be complete. The information in Table 9-15 is presented to provide a snap-shot of the state of groundwater investigations in the area of the LDW. Concentration ranges for selected chemicals detected in groundwater, seeps, and porewater are provided in Appendix I, as available, for many of the facilities listed in Table 9-15.[154] In-depth analyses have not been conducted as part of the RI for any of the facilities in Table 9-15; however, a preliminary assessment of 12 facilities was conducted as part of the groundwater pathway assessment in 2003 (Appendix G in Windward 2003a), based on data available at the time. Groundwater

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[152] The RI seep survey(Windward 2004b) was conducted over the entire LDW. In consultation with EPA and Ecology, some of the seeps observed were selected for sampling during the seep study. The RI porewater study (Windward 2006a) was conducted near the Great Western International and Boeing Plant 2/Jorgensen Forge properties because these properties were identified as having the greatest potential for VOC-contaminated porewater as a result of groundwater discharge, based on information available at that time.

[153] For facilities that were not included in the Phase 1 RI groundwater pathways assessment, the administrative SCA sediment boundaries provided by Ecology in February 2007 (Ecology 2007j) were used to produce the list of chemicals detected above the SQS. Those boundaries were established only for administrative purposes and are based on the extent of the adjacent upland LDW sub-drainage basins of the SCAs; the boundaries are not intended to delineate potential sediment cleanup boundaries.

[154] Additional information is provided in Appendix I for facilities located within one of the 11 SCAs selected for summation.

pathway analysis is dependent on an assessment of a number of factors, including site-specific hydrogeology characteristics and chemical-specific fate and transport parameters. EPA and Ecology will continue to evaluate chemicals in groundwater and seeps as part of their continued source control efforts. Information on the hydrogeology of the LDW, including geology, physical properties, and groundwater systems, is included in Section 2.5.

Section 9.4.6.1 Phase 1 groundwater pathway assessment

The groundwater pathway assessment summarized information available through 2002 for 12 upland facilities identified by EPA and Ecology as preliminary sites of interest with respect to groundwater. These facilities were Advance Electroplating, Boeing Developmental Center, Boeing Isaacson, Boeing Plant 2, GWI, Long Painting, T-117 (former Malarkey Asphalt), PACCAR (former Kenworth Truck Co.), Philip Services/Burlington Environmental, the former Rhône-Poulenc facility, the South Park Landfill, and T-108 (former Chiyoda property) (Appendix G in Windward 2003a). The locations of these facilities are shown on Map 9-17.

The following information was evaluated in the groundwater pathway assessment for each of the 12 facilities:

- Site operations and regulatory status
- Site-specific aquifer characteristics, groundwater flow direction, and rates of flow
- Chemicals identified as groundwater COCs (different criteria were used to select COCs for different facilities)
- COC concentrations in groundwater samples collected from monitoring wells located nearest to the LDW, as well as chemical contour or plume maps, when available
- COC concentrations in groundwater seeps or sediment in the LDW near potential areas of groundwater discharge
- Available fate and transport information

Based on various screening criteria used at each of the facilities, COCs identified in groundwater included chlorinated solvents and their breakdown products, metals, BTEX compounds, TPH, PCBs, PAHs, and a few other organic compounds at a few facilities. The most common groundwater COCs were metals and chlorinated solvents, both of which were COCs at over half of the facilities. Groundwater COC lists for individual facilities may be updated by EPA or Ecology based on ongoing environmental investigations and source control work.

As part of the groundwater pathway assessment (Appendix G in Windward 2003a), zones of potential groundwater discharge were estimated for each of the 12 facilities, and surface sediment data available at that time within that zone were reviewed to identify detections of groundwater COCs in sediment at concentrations above SMS criteria or DMMP guidelines.[155] This comparison of sediment data to the SQS criteria or DMMP guidelines is intended only as a preliminary assessment; chemicals in groundwater may also pose a risk to sediment-dwelling organisms through porewater exposure. The groundwater discharge zones were estimated based on the direction of groundwater flow from each facility to the LDW; additional investigations would be necessary to formally delineate the zone of potential groundwater discharge for each facility. Additional source analyses may be conducted prior to remediation for specific facilities as part of cleanup-related activities.

Limitations of the groundwater pathway assessment were associated with the type of data used for the analysis and uncertainties associated with the data. Uncertainties included: 1) all potential sources of groundwater contamination, such as the presence of uncharacterized fill material, may not have been addressed; 2) few groundwater samples were available for facilities on the west side of the LDW; and 3) SMS criteria or DMMP guidelines were unavailable for some chemicals. The preliminary results provided in the groundwater pathway assessment for each of the 12 facilities are summarized below and have been updated based on groundwater, seep, and porewater information collected or identified since completion of the assessment in 2003. Groundwater information is still being collected at several facilities in the LDW drainage basin and reviewed by EPA and Ecology. EPA and Ecology will make the final pathway determinations based on their continuing source control evaluations.

9.4.6.2 Seep study conducted for the LDW RI

In 2004, a seep study was conducted as part of the RI to evaluate whether seeps contributed significant chemical inputs to the LDW, to determine whether additional seeps should be sampled in the future as part of the RI or to provide source control information, and to determine whether additional surface sediment sampling was needed in seep areas (Windward 2004b). A reconnaissance survey that resulted in the visual identification of 82 seep locations throughout the LDW was conducted; 18 of these locations were selected in consultation with EPA and Ecology for sample collection and chemical analysis. Seeps were selected for analysis based on the seep’s proximity to potentially contaminated upland properties according to groundwater information available at the time, seep flow, and any visual or olfactory indications (e.g., oily or colored seep water or intertidal sediment) that might suggest the potential presence of chemicals in the seep. Two of the 18 seeps selected for chemical analysis were not sampled because they were dry at the time of sampling. The locations of the 16 seeps sampled during the LDW seep study are shown on Map 4-11a.

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[155] Sediment information provided in Table 9-15 and discussed in this section is based on the RI baseline surface sediment dataset, which includes sediment data collected since the groundwater pathway assessment was completed.

Seeps have also been sampled as part of several other investigations in the LDW. Information on these sampling events is provided in Section 4.1.2.5 (Table 4-11 and Map 4-11a). The results of the seep sampling conducted for the RI and as part of other investigations are presented in multiple places throughout the RI: Maps 4-11b through 4-11e, Section 4.2 (where seep results are described in detail by chemical group), Table 4-21 (where seep data are compared with WQC), Table 9-15 (where chemicals detected in the seep samples are listed by facility), and the following sections where seep data associated with the 12 facilities included in the groundwater pathway assessment are discussed. Seep data are also presented in Appendix I for the facilities located within the 11 SCAs selected for summarization. EPA and Ecology may further evaluate seeps as part of their continuing source control efforts.

9.4.6.3 Porewater study conducted for the LDW RI

A porewater study was conducted as part of the RI in 2005 to determine whether VOCs in porewater posed a risk to benthic invertebrates (Windward 2006a). Although VOCs rarely accumulate in sediment, organisms such as benthic invertebrates could be exposed to VOCs in areas downgradient from upland areas with high VOC concentrations in groundwater. Porewater samples were collected from areas adjacent to upland properties where VOCs had historically been detected in groundwater (Map 4-12a). The GWI and Boeing Plant 2/Jorgensen Forge facilities were selected for the porewater investigation because existing data indicated that these areas had the greatest potential to have VOC-contaminated porewater as a result of groundwater discharge (i.e., maximum concentrations of some VOCs were at least an order of magnitude higher at these facilities than at any of the other 12 facilities evaluated[156] (Windward 2005h)).

Porewater sampling has also been conducted as part of several other investigations in the LDW. Information on porewater sampling events is provided in Section 4.1.2.6 (Table 4-12 and Map 4-12a). The results of the porewater sampling conducted for the RI and as part of other investigations are presented in multiple places throughout the RI: Maps 4-12b through4-12d, Section 4.2 (where porewater results are described in further detail by chemical group), Table 4-22 (where porewater data are compared with WQC), Table 9-15 (where chemicals detected in the porewater samples are listed by facility), and the following sections where porewater data associated with the 12 facilities included in the groundwater pathway assessment are discussed. Porewater data are also presented in Appendix I for the facilities located within the 11 SCAs

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[156] One possible exception was trichloroethene at Advance Electroplating, according to EPA's summary of other data collected from seven onsite wells by Hart Crowser in 2000 (Sanga 2002). The high concentrations of trichloroethene (200,000 µg/L) in one of the site wells suggest that non-aqueous phase liquid (NAPL) may be present beneath the site, although it was not detected in the 104 borings drilled to 13 ft by Ecology & Environment, or in the seven monitoring wells installed by Hart Crowser (Sanga 2002). The highest trichloroethene concentration documented in follow-up work was 2,600 µg/L.

selected for summarization. Ecology and the SCWG may further evaluate chemicals in porewater as part of their continued source control efforts.

9.4.6.4 Groundwater, seep, and porewater study information for the 12 facilities included in the Phase 1 groundwater pathway assessment

This section presents a summary of groundwater, seep, and porewater information available for each of the 12 facilities evaluated in the groundwater pathway assessment (Appendix G in Windward 2003a). Information collected or identified since the completion of that assessment is also included.

Advance Electroplating

Advance Electroplating is located approximately 3,300 ft to the west of the LDW at RM 4.1 (Map 9-17). The facility was used for chrome plating operations from the mid-1960s to the early 1990s. A time-critical removal action (TCRA) under RCRA was conducted at the facility in the mid-1990s (Appendix G in Windward 2003a). Chemicals identified as groundwater COCs that were evaluated in the groundwater pathway assessment included 10 metals (antimony, arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, and zinc) and 2 VOCs (tetrachloroethene and trichloroethene).[157]

The zone of potential groundwater discharge for Advance Electroplating was estimated to be between RM 4.1 and RM 4.2 (on the west side of the LDW) (Appendix G in Windward 2003a); however, groundwater discharge zones for this facility have not been formally delineated. This zone was identified because it is the area where Hamm Creek discharges; it is probable that the Advance Electroplating facility's shallow groundwater discharges to a local ditch or other surface water drainage (e.g., Hamm Creek) before discharging to the LDW, given the relative elevation difference between the site and the LDW elevation, and the surface water drainage systems between the site and the LDW (Appendix G in Windward 2003a). Additional groundwater investigations would be needed to formally delineate the groundwater discharge zone for this facility.

Water quality data are available for Hamm Creek (Herrera 2004b, 2005) and were used in a pollutant loading analysis conducted for the Green-Duwamish watershed (Herrera 2007). The pollutant loading analysis included a calculation of loading values for Hamm Creek for total and dissolved copper, total and dissolved mercury, and total and dissolved zinc. This area was also included in the lateral loads analysis discussed in Sections 3.2.5.1 and 9.4.4.6. Based on a review of the baseline surface sediment data collected from the zone of potential groundwater discharge, none of the potential

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[157] Chemicals were selected as groundwater COCs for Advance Electroplating in the groundwater pathway assessment if they were detected in groundwater and if SMS criteria or DMMP guidelines were available (Windward 2003a).

groundwater COCs identified for the Advance Electroplating facility were detected above the SQS.[158]

Two porewater samples (unfiltered) were collected from the zone of potential groundwater discharge as part of an EPA SI (Weston 1999a) (Map 9-17). Arsenic, barium, lead, manganese, and vanadium were detected in both porewater samples; copper was detected in one porewater sample. All of these chemicals were also detected in site groundwater (Table 9-15). VOCs were not analyzed in the porewater samples. No seep data from the zone of potential groundwater discharge were identified.

The groundwater quality within the deeper alluvium has not been evaluated. The clay layer identified above the alluvium by Ecology and Environment (1997) may have restricted downward COC migration. EPA's emergency action removed most of the contaminated soils from the facility in 1995 and 1996 (Ecology and Environment 1997).

Boeing Developmental Center

The BDC is located on the east side of the LDW between RM 4.2 and RM 4.8 (Map 9-17). One hundred and fifty-seven solid waste management units (SWMUs) and five areas of concern (AOC) were identified and investigated at the facility under RCRA (Ecology and Environment 2007a). All but two of the SWMUs (SWMU-17, SWMU-20) were determined to not represent a threat to human health or the environment and were clean-closed in accordance with RCRA requirements (Appendix G in Windward 2003a; Ecology and Environment 2007a). Of the SWMUs and AOCs that required further monitoring, three areas required additional groundwater investigation: SWMU-17, SWMU-20, and AOC 05. These areas are all located on the southern portion of the BDC.

Chemicals detected in groundwater at the BDC that were evaluated in the groundwater pathway assessment were arsenic, copper, lead, nickel, benzene, tetrachloroethene, and TPH.[159] The zone of potential groundwater discharge for the BDC was estimated to be the east side of the LDW between RM 4.4 and RM 4.8 (Appendix G in Windward 2003a); however, groundwater discharge zones for this facility have not been formally delineated. Of the groundwater COCs for which SMS criteria exist, only lead was detected in surface sediments above the SQS (i.e., one CSL exceedance was detected near the southern end of the zone of potential groundwater

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[158] All of the potential groundwater COC metals were analyzed for in all of the surface sediment samples collected from the estimated groundwater discharge zone; the two VOCs were only analyzed in one sample.

[159] Chemicals were selected as groundwater COCs for the BDC in the groundwater pathway assessment if they had been detected since 1998 in groundwater above MTCA Method B marine surface water cleanup levels (Appendix G in Windward 2003a).

discharge). No seep or porewater data have been collected from within the zone of potential groundwater discharge for this facility.

Remedial actions at SWMU-17, SWMU-20, and AOC 05 have been conducted under RCRA corrective action authority. Pump and treat groundwater remediation was conducted at SWMU-20 from 1993 to 2001, and an electron donor injection was performed in 2004.[160] Groundwater monitoring conducted in May 2006 indicated that chemical concentrations at SWMU-20 have decreased since the initiation of groundwater treatment (Ecology and Environment 2007a). The groundwater monitoring data from SWMU-20 showed that concentrations of chlorinated VOCs decreased downgradient of the former degreaser pit and that concentrations also decreased with depth so that the downgradient extent of chlorinated solvent detection was limited to within 200 ft of the original source area. The LDW is another 500 ft from the downgradient extent of the detected VOC plume (Appendix G in Windward 2003a).

At AOC 05, an oxygen release compound was injected into groundwater in 2002, and pilot testing of a bioremediation groundwater remediation system was initiated in 2007 (Ecology and Environment 2007a) to address TPH and benzene contamination.[161] Groundwater monitoring is ongoing at SWMU-17, SWMU-20, and AOC 05.

The data reviewed for the BDC as part of the groundwater pathways assessment indicated that the residual COC plumes are contained locally around the source areas. In the case of AOC 05, the UST source has been removed (Landau 2001), and the existing monitoring data indicate a very limited extent of residual hydrocarbon concentrations in groundwater (Appendix G in Windward 2003a). Given the ongoing cleanup activities at SWMU-20 and AOC 05, and the distance from SWMU-17, SWMU-20, and AOC 05 to the LDW (over 700 ft), attenuation through biodegradation could be expected to reduce the residual hydrocarbon constituent concentrations before discharge to the LDW.

Additional information on the BDC is included in both the Slip 6 and Norfolk CSO/SD SCA summaries in Appendix I (Sections I.4.7 and I.4.8).

Boeing Isaacson

The Boeing Isaacson facility is located on the east side of the LDW between RM 3.7 and RM 3.8 (Map 9-17). Several remedial actions and environmental investigations have been conducted at the facility (Appendix G in Windward 2003a) (also see

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[160] SWMU-20 is approximately 0.25 ac in size.

[161] Toluene, enthylbenzene, and xylene are being remediated at AOC 05 by oxygen release compound injection and the bioremediation system.

Appendix I). Chemicals detected in Boeing Isaacson groundwater that were evaluated in the groundwater pathway assessment were arsenic, chromium, lead, and zinc.[162]

The zone of potential groundwater discharge for Boeing Isaacson was estimated to be between RM 3.7 and RM 3.9 on the east side of the LDW (Appendix G in Windward 2003a); however, groundwater discharge zones for this facility have not been formally delineated. With the exception of arsenic, none of the chemicals identified in the groundwater pathway assessment were detected above the SQS in surface sediments in this area. Arsenic was detected above the CSL in two samples and above the SQS in one sample collected within the zone of potential groundwater discharge (see Map 4-14e and Appendix I, Section I.4.6).

One seep sample was collected in support of a request for a groundwater NFA determination at the Boeing Isaacson facility (ERM and Exponent 2000). Dissolved arsenic was detected in the sample. An additional seep location (SP-20) along the Jorgensen Forge shoreline near the Jorgensen Forge and Boeing Isaacson property boundary was sampled in 2004 (Windward 2004b). Arsenic, cadmium, copper, lead, mercury, nickel, silver, and zinc were detected in both filtered and unfiltered samples collected from this seep location.

Additional sampling was conducted in 2006 and 2007, and samples were analyzed for total and dissolved arsenic (SAIC 2008b). Data from these sampling events are summarized in Appendix I (Section I.4.6).

A comparison of arsenic concentrations in groundwater samples collected in the 1990s near the shoreline relative to samples collected from upgradient wells located slightly inland from the shoreline identified an apparent attenuation process occurring in groundwater at Boeing Isaacson (Appendix G in Windward 2003a). These wells are screened within the same aquifer zone and indicated at least an order of magnitude reduction in arsenic concentrations within 100 ft of downgradient flow (Appendix G in Windward 2003a). The potential for arsenic in groundwater to contaminate LDW sediments is being investigated at this site by Ecology (Ecology 2008a). Groundwater sampling was conducted in 2007 and additional sampling is planned (Ecology 2008a).

Ecology is still assessing groundwater at this facility as part of their source control program for the LDW. Ecology is negotiating an Agreed Order with Boeing to conduct a MTCA RI/FS at the Boeing Isaacson site and the adjacent Boeing Thompson site. Under this order, Boeing will conduct: 1) additional soil sampling on these properties to characterize concentrations of arsenic and other COCs in surface and subsurface soils, and 2) additional groundwater sampling at selected wells on these properties during both wet and dry seasons to characterize concentrations of arsenic and other

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[162] Chemicals were selected as groundwater COCs for Boeing Isaacson in the groundwater pathway assessment if they were detected in the 1988 sampling event above either their freshwater or marine AWQC (Appendix G in Windward 2003a).

COCs in groundwater. If COCs in soil or groundwater are present at sufficiently high concentrations, and there is demonstrated to be a pathway to the LDW that could result in unacceptable levels of sediment contamination, Boeing will develop a plan for controlling these contaminant sources.

Boeing Plant 2

The Boeing Plant 2 facility is located on the east side of the LDW between approximately RM 2.9 and RM 3.6 (Map 9-17). Environmental investigations and corrective actions have been conducted at Boeing Plant 2 under RCRA since 1994 (see Appendix I, Section I.4.5). Groundwater data collected as part of several site investigations (Weston 1996, 1998, 2001a, b, c, 2002a; Weston Solutions 2002) were evaluated in the groundwater pathway assessment (Appendix G in Windward 2003a). Chemicals that were identified as groundwater COCs in the groundwater pathway assessment were arsenic, cadmium, chromium, copper, lead, nickel, selenium, silver, thallium, zinc, PCBs, 1,1-dichloroethene, cis-1,2-dichloroethene, trichloroethene, and vinyl chloride.[163] Shoreline groundwater monitoring at Boeing Plant 2 is ongoing. Nine VOCs, thirteen metals, and PCBs were detected in samples from a recent semi-annual monitoring event (Environmental Partners 2008). Data gaps investigations have also recently been completed for upland areas at Boeing Plant 2 (Environmental Partners and Golder Associates 2007a, c, b; Ecology and Environment 2007b). Recent data are presented in Appendix I.

Based on groundwater flow information, the zone of potential groundwater discharge for Boeing Plant 2 was estimated as extending from RM 2.8 to RM 3.6 on the east side of the LDW (Appendix G in Windward 2003a); however, groundwater discharge zones for this facility have not been formally delineated. A review of the baseline surface sediment dataset (see Appendix I, Section I.4.5) indicates that of the groundwater COCs for which SMS criteria are available,[164] PCBs and all of the metals except for arsenic were detected in some areas at concentrations above the CSL in the zone of potential groundwater discharge.

In 1995, 21 seep samples (some filtered and some unfiltered) were collected from 18 locations along the Boeing Plant 2 and adjacent Jorgensen Forge facility shoreline as part of the Plant 2 RFI (Weston 1998) (see Map 4-11a). Of the VOCs identified as groundwater COCs in the groundwater pathway assessment (Appendix G in Windward 2003a), cis-1,2-dichloroethene, trichloroethene, vinyl chloride, copper (total and dissolved), zinc (total), and total PCBs were detected in the seep samples.

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[163] The chemicals selected in the groundwater pathway assessment as the groundwater COCs for the Boeing Plant 2 facility were those detected in boundary wells above preliminary media cleanup levels (Appendix G in Windward 2003a).

[164] SMS criteria are not available for nickel, selenium, thallium, 1,1-dichloroethene, cis-1,2-dichloroethene, or vinyl chloride.

In 2005, as part of the LDW RI (Windward 2006a), 14 porewater samples were collected from locations adjacent to the Boeing Plant 2 and Jorgensen Forge facilities (see Map 4-12a) and analyzed for VOCs. Four VOCs (vinyl chloride, 1,1-dichloroethene, cis-1,2-dichloroethene, and trichloroethene) were detected in porewater samples collected using peepers. At least one VOC was detected in samples collected from five of the eight locations at this site, with concentrations ranging from 0.2 µg/L (cis-1,2-dichloroethene and trichloroethene) to 13 µg/L (vinyl chloride).

A roughly 15-ac area of intertidal and subtidal sediments (referred to as the Duwamish Sediments Other Area [DSOA]) along the Plant 2 shoreline has been delineated for dredging and backfilling/capping remedial actions (Ecology and Environment 2007b) (see Appendix I, Section I.4.5). The approval and design of the dredging and backfilling/capping interim measure are currently in progress (Ecology and SAIC 2008). Further investigation is being planned to more fully characterize the depth of sediment contamination adjoining the navigation channel. As part of the planning for sediment remediation, Weston conducted a mass loading analysis and an equilibrium-partitioning model analysis to evaluate whether metals in groundwater had the potential to recontaminate sediment in the remediation area (Weston 2002b). The analyses indicated that after remediation, groundwater from the facility would not contaminate the sediment area at concentrations that would exceed the SQS within a reasonable time frame (i.e., hundreds of years). EPA has not expressed an opinion on the conclusions reached in any of Boeing's reports regarding groundwater as a source of contaminants to the LDW, and EPA has not yet decided whether there will be any need for further control of groundwater.

Great Western International

GWI is located on the east side of the LDW approximately 400 ft to the northeast of the Myrtle Street embayment at RM 2.4 (Map 9-17). An RI/FS was initiated at the facility in 1991 to address soil and groundwater contamination associated with site operations (Appendix G in Windward 2003a). In 2004, an underground injection control cleanup program was implemented. As of 2006, cleanup was still being conducted and its effectiveness was still under evaluation (Ecology and Environment 2008a). The groundwater pathway assessment used groundwater data obtained from the supplemental remedial investigation (SRI)/FS (Terra Vac and Floyd & Snider 2000). Chemicals that were identified as groundwater COCs were tetrachloroethene, trichloroethene, cis-1,2-dichloroethene, trans-1,2-dichloroethene, vinyl chloride, 1,1-dichloroethane, and methylene chloride.[165]

Based on groundwater flow information, the zone of potential groundwater discharge for GWI was estimated to be between RM 2.3 and RM 2.4 (Appendix G in Windward

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[165] A chemical was identified in the SRI/FS as a COC for GWI if it was detected in monitoring wells located closest to the LDW above MTCA Method B cleanup levels.

2003a); however, the groundwater discharge zone for this facility has not been formally delineated. One of the four surface sediment samples collected from within the zone of potential groundwater discharge was analyzed for VOCs, including all of the groundwater COCs, except methylene chloride. None of the groundwater COCs that were analyzed for in sediment were detected.

As part of the SRI/FS, 43 water samples were collected from sediment in the Myrtle Street embayment using absorbent screening modules called Gore-Sorbers® (Terra Vac and Floyd & Snider 2000). Two of the water samples contained detectable quantities of tetrachloroethene breakdown products. Seep samples were also collected in 1998 and 1999 as part of the SRI/FS (see Map 4-11a); it is not known whether the seep samples were filtered or unfiltered. All of the groundwater COCs, with the exception of methylene chloride, were detected in the seep samples.

The key findings of the Myrtle Street embayment study (Terra Vac and Floyd & Snider 2000) were as follows:

- Groundwater discharge to the LDW at the Myrtle Street embayment was found to occur as discrete seep discharges in the intertidal zone. Generalized groundwater upwelling was not observed in the Myrtle Street embayment based on the Gore-Sorber study using VOCs as chemical markers.
- PCE and/or its degradation products were detected in Gore-Sorber modules placed in seep-face sediments along the eastern edge of the Myrtle Street embayment. This seep face was further delineated in a second sampling event and found to have a moderately long horizontal extent. The seeps were bounded in elevation (vertical extent) and found to occur between approximately -1 and -3 ft MLLW.

Porewater samples were collected from the zone of potential groundwater discharge for GWI as part of the RI (Windward 2006a) (see Map 4-12a). Piezometers were used to collect porewater samples in deeper areas, and peepers (porous diffusion samplers) were used in shallower areas. The samples were analyzed for all of the groundwater COCs identified for GWI, with the exception of methylene chloride. None of the groundwater COCs were detected in porewater samples collected using piezometers, consistent with the results of the Gore-Sorber study and the conceptual site model (Windward 2005h). In samples collected using peepers, all of the groundwater COCs analyzed were detected in two or more samples. Cis-1,2-dichloroethene and vinyl chloride were detected in every peeper sample collected from the zone of potential groundwater discharge for GWI. The VOC results for porewater are discussed further in Section 4.2.11.4.

Based on groundwater, seep, Gore-Sorber®, and surface sediment data available at the time, the SRI/FS (Terra Vac and Floyd & Snider 2000) presented a fate and transport analysis of ethenes and ethanes. That analysis indicated that groundwater COCs were extensively degraded through reductive dechlorination.

Ecology is continuing to assess the GWI facility under MTCA and as part of their source control program for the LDW (SAIC 2008a; Ecology 2009). Ecology and the GWI property owner (the facility is now called the Fox Avenue Building LLC site) are entering into an Agreed Order under MTCA for additional site investigation and cleanup. An interim cleanup action to address contaminant migration to the LDW is included in the Scope of Work under the Agreed Order for GWI.

Long Painting

The Long Painting facility is located on the west side of the LDW between approximately RM 3.0 and RM 3.1 (Map 9-17). A portion of the facility is adjacent to the LDW, and a portion is inland. Site investigations conducted in 1997 and 2000 (Kleinfelder 2000) included groundwater sampling (Appendix G in Windward 2003a). Groundwater data collected in 2000 were used in the groundwater pathway assessment. Chemicals in groundwater that were identified as COCs were arsenic, chromium, lead, 1,1,1-trichloroethane, and tetrachloroethene.[166]

The zone of potential groundwater discharge for Long Painting was estimated to be between RM 2.9 and RM 3.1 on the southwest side of the LDW, based on groundwater flow information (Appendix G in Windward 2003a); however, the groundwater discharge zone for this facility has not been formally delineated. None of the metals were detected above the SQS in the surface sediment samples collected from the zone of potential groundwater discharge in which they were analyzed (see Map 4-14d). The VOCs were not detected in the single sediment sample analyzed in this zone (Appendix G in Windward 2003a).

One porewater sample was collected just outside the zone of potential groundwater discharge for Long Painting; this sample was collected within the navigation channel at approximately RM 3.05 as part of the PSDDA sediment characterization of the LDW navigation channel (SEA 1998). None of the groundwater COCs for Long Painting were analyzed in the porewater sample.

One seep sample was collected from the zone of potential groundwater discharge as part of the RI (Windward 2004b). Of the groundwater COCs for this facility, arsenic and lead were detected in the seep sample. Neither 1,1,1-trichloroethane nor tetrachloroethene was analyzed in this sample.

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[166] The metals identified as COCs were selected in the groundwater pathway assessment for the Long Painting facility because they were detected in groundwater above MTCA Method A or B groundwater cleanup levels and in the monitoring well closest to the LDW (Appendix G in Windward 2003a). The two VOCs were not detected above MTCA cleanup levels in any of the wells but were selected because it was not clear if the downgradient monitoring wells would capture a potential chlorinated solvent plume.

The groundwater pathway assessment concluded that there were insufficient groundwater data (i.e., representation of depth and location) to fully evaluate the potential for groundwater contamination on the Long Painting site.

T-117 (Former Malarkey Asphalt)

T-117 is the location of the former Malarkey Asphalt facility, which manufactured roofing asphalt and other petroleum-based products from 1937 to 1993. The site is located on the west side of the LDW, between approximately RM 3.5 and RM 3.7 (Map 9-17). Several environmental investigations and remedial actions have been conducted at the T-117 facility; and a non-time-critical removal action (NTCRA) for the remediation of soil, sediment, and adjacent streets is planned. Additional information on the background of and plans for the T-117 facility is included in the groundwater pathway assessment (Appendix G in Windward 2003a) and Appendix I, Section I.4.11.

Groundwater data were collected from the site between 2003 and 2008 as part of the draft EE/CA for the NTCRA (Windward et al. 2008). Total PCBs, TPH, and chrysene were selected as the groundwater COCs for the T-117 facility in the EE/CA (Windward et al. 2008). As stated in the draft EE/CA, these groundwater COCs are expected to be addressed through the NTCRA removal of contaminated soil, which is the likely source of contamination to the groundwater (Windward et al. 2008).

The zone of potential groundwater discharge for the T-117/former Malarkey Asphalt facility was estimated to be the sediment area adjacent to the site, extending from RM 3.5 to RM 3.7, based on the proximity of the facility to this area and groundwater flow information (Appendix G in Windward 2003a); however, the groundwater discharge zone for this facility has not been formally delineated. PCBs and several individual PAHs have been detected above the CSL in numerous sediment samples collected from the zone of potential groundwater discharge (see Appendix I, Section I.4.11).

In 2003, seep samples were collected from three locations within the zone of potential groundwater discharge for T-117 as part of the NTCRA investigation (Windward et al. 2005a). Total PCBs were detected in one of the three seep samples. This seep was re-sampled because it was suspected that the detected PCB concentration of 0.94 J µg/L may have been associated with suspended solids in the water sample. The second sample was centrifuged and analyzed, and PCBs were not detected at a reporting limit of 0.033 µg/L. No other groundwater COCs were detected in the seep samples. No porewater data have been collected.

Groundwater data have been collected at T-117 since 1991. Historical (pre-2003) groundwater conditions are summarized in the data gaps report (Windward et al. 2003). Groundwater monitoring has been conducted at T-117 periodically since 2003. Nine wells are currently located on the upland portion of the T-117 facility. Six wells are located along the shoreline (25 to 70 ft from the LDW), and three wells are upgradient (ENSR|AECOM 2008). Recent groundwater data for T-117 are summarized in Appendix I (Section I.4.11).

PCBs have been detected in 11 out of 35 unfiltered groundwater samples collected between 2005 and 2008 in shoreline and upgradient wells at concentrations ranging from 0.010 to 2.0 µg/L (Windward et al. 2005a; 2008). TPH was detected in 4 out of 19 samples ranging from 0.5 to 7.5 mg/L. Chrysene was detected in one sample at 0.50 µg/L.

In 2004, five shoreline wells were sampled. PCBs and chrysene were not detected. Samples collected from shoreline wells in 2005 were analyzed for PCBs and TPH. PCBs were detected in one shoreline well at a concentration of 0.040 µg/L, and TPH was detected in another shoreline well at a concentration of 0.5 mg/L. In January 2006, the well with the PCB detection was re-sampled to confirm the 2005 result. PCBs were detected in the 2006 sample at 0.32 µg/L.

In August 2006, six shoreline wells were sampled. PCBs were detected in four of the shoreline wells, TPH was detected in one shoreline well at 0.94 mg/L, and no PAHs were detected.

In March 2008, groundwater samples were collected from the six shoreline wells and the three upgradient wells. PCBs were detected in four wells and TPH and PAHs were detected in two wells. TPH concentrations were 0.70 mg/L in a shoreline well and 7.5 mg/L in an upgradient well. Chrysene was detected in one well. Groundwater monitoring is still being conducted at T-117 as part of a quarterly groundwater monitoring investigation.

Although PCBs were detected in shoreline wells, PCBs have very low solubility and mobility in groundwater, except in the presence of a co-solvent such as petroleum fuel. Trace amounts of LNAPL (i.e., a sheen < 0.01 ft thick) were observed in two of the monitoring wells (MW-2 and MW-7) on one occasion during the 2005 tidal study. During the most recent groundwater monitoring event in February 2008 (ENSR|AECOM 2008), no LNAPL was observed in any of the wells monitored. Concentrations of petroleum hydrocarbons (TPH-oil) were detected during quarterly groundwater sampling.

Groundwater in these wells is typically turbid, indicating the likelihood that low concentrations of PCBs present in the fine silts surrounding the well screens may be leaching into the wells during well purging. Thus, PCB detections in the groundwater were likely the result of suspended soil particles (Windward et al. 2008).

As stated in the draft EE/CA for the upland NTCRA, collectively, these data suggest that a majority of the PCBs detected in sediment near T-117 are most likely the result of past releases of surface water runoff, spillage from the upland area, or erosion of contaminated soil in the shoreline bank(Windward et al. 2008).[167] Seep monitoring data collected to date indicate that seep discharges to the LDW do not appear to be a source to the sediment because chemicals detected in the seep samples (metals and BEHP) do not exceed the SQS in sediment and were not identified as site COCs (Windward et al. 2008). PCBs and chrysene were not detected in the seep samples (after centrifugation). However, PCBs have been detected in groundwater samples. The draft EE/CA (Windward et al. 2008) for the upland NTCRA is under review by Ecology. The goal of the upland cleanup is to remove most of the contamination from the upland site.

PACCAR (Former Kenworth Truck Co.)

The PACCAR facility (a former Kenworth Truck Co. manufacturing facility) is located on the east side of the LDW between RM 3.9 and RM 4.0 (Map 9-17). Environmental investigations and remedial activities, including groundwater extraction, were conducted at the facility in the 1990s in association with contamination from the former USTs located at the facility (see Appendix I, Section I.4.7). Groundwater data collected in the late 1990s and early 2000s (Kennedy/Jenks 2002; GeoEngineers and Kennedy/Jenks 1990; Kennedy/Jenks 1996, 1999) were evaluated in the groundwater pathway assessment (Appendix G in Windward 2003a). Chemicals in groundwater that were identified as COCs were arsenic, barium, 1,1-dichloroethene, trichloroethene, tetrachloroethene, and vinyl chloride.[168]

The zone of potential groundwater discharge for the PACCAR/former Kenworth Truck Co. facility was estimated to be the sediment area adjacent to the site, extending from RM 3.9 to RM 4.0, based on the proximity of the facility to this area and groundwater flow information (Appendix G in Windward 2003a); however, the groundwater discharge zone for this facility has not been formally delineated. Arsenic was not detected in surface sediments at concentrations greater than the SQS in the zone of potential groundwater discharge (see Map 4-14e). Barium was detected in three sediment samples collected from the zone of potential groundwater discharge; there are no SMS criteria or DMMP guidelines for barium. None of the VOCs identified as groundwater COCs have been analyzed in surface sediment samples collected from the zone of potential groundwater discharge.

Five seeps were sampled as part of a data gaps investigation at the PACCAR facility in 2002 (Ecology and Environment 2008b). Total arsenic was detected in one of the seep

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[167] Any statements referring to the draft EE/CA should not be considered as the final opinion of EPA and Ecology.

[168] The groundwater COCs identified in the groundwater pathway assessment for the PACCAR/former Kenworth Truck Co. facility were those chemicals that were detected above MTCA Method B or C cleanup levels or detected at concentrations greater than the cleanup and remediation levels at the adjacent Boeing Thompson facility (as reported in Kennedy/Jenks 2002; Appendix G in Windward 2003a).

samples. Four seeps were sampled again in 2007; dissolved copper was detected (Ecology and Environment 2008b). The data gaps report (Ecology and Environment 2008b) did not specify whether barium or VOCs were analyzed in the seep samples collected in 2002 and 2007.

The VOCs identified as COCs in site groundwater are associated with past releases from leaking USTs (Appendix G in Windward 2003a). Tank removal and pump and treat remediation activities were conducted in this area. Groundwater monitoring data showed improvement in the groundwater quality during remediation (Appendix G in Windward 2003a). In addition, an air sparging/soil vapor extraction (AS/SVE) system has been used at the facility since 2004 to treat a VOC plume in groundwater on the northwest portion of the facility (Ecology and Environment 2008b). The AS/SVE system consists of over 30 air sparging wells and 6 soil vapor extraction lines.

Site-wide groundwater monitoring was conducted in 2004 as part of the Phase II data gaps investigation at the facility (Ecology and Environment 2008b). VOCs were detected in groundwater on the western portion of the facility, and arsenic was detected above natural background. Additional groundwater monitoring was conducted in 2006 as part of a wet- and dry-season groundwater study. Vinyl chloride was identified by Ecology as a potential COC based on groundwater sampling results in nearshore wells; however, arsenic was not identified as a groundwater COC at the facility (Ecology and Environment 2008b).

Groundwater monitoring was also conducted in 2007 to assess the effectiveness of the AS/SVE system at treating the VOC plume. Wells to the north, south, and downgradient of the AS/SVE system (adjacent to the LDW) were sampled. Sampling confirmed that VOCs were below MTCA Method B surface water cleanup levels (Ecology and Environment 2008b). Recent groundwater data are summarized in Appendix I (Section I.4.7).

Ecology is still assessing groundwater at this facility as part of two Agreed Orders signed with PACCAR and the property owner (Merrill Creek Holdings LLC) and as part of their source control program for the LDW.

Philip Services /Burlington Environmental

The Philip Services/Burlington Environmental facility is located approximately 4,300 ft to the east of the LDW at RM 1.4 (Map 9-17). The facility was entered into Ecology's CSCSL on March 1, 1988, and is listed as having confirmed groundwater, soil, and air contamination (Ecology 2007l). Confirmed contaminants in groundwater are halogenated organic compounds, EPA priority pollutants (metals and cyanide), PCBs, petroleum products, phenolic compounds, and PAHs. Suspected contaminants in groundwater include base/neutral/acid organics, non-halogenated solvents, and arsenic. Contaminants in groundwater listed as below MTCA cleanup levels after assessment include pesticides, reactive wastes, corrosive wastes, radioactive wastes, asbestos, and methyl tertiary butyl ether. Confirmed contaminants in soil are identified as base/neutral/acid organics, halogenated organic compounds, EPA priority pollutants (metals and cyanide), PCBs, phenolic compounds, PAHs, and arsenic. An RFI was initiated in the late 1980s. Several investigations into groundwater conditions and potential sources of groundwater contamination were completed for this facility as part of the RFI and subsequent groundwater monitoring. Groundwater investigations conducted by Philip Services at offsite locations near the facility indicated there were likely additional sources of chlorinated solvents between the facility and the LDW.

Groundwater data evaluated as part of the groundwater pathway assessment (Appendix G in Windward 2003a) were collected from wells thought to represent the western boundary of the groundwater plume originating from the Philip Services/Burlington Environmental facility. Groundwater COCs that were selected for evaluation in the groundwater pathway assessment were benzoic acid, barium, cadmium, chromium, copper, lead, manganese, nickel, silver, 1,1,1-trichloroethane, 1,1-dichloroethane, chloroethane, 1,1-dichloroethene, cis-1,2-dichloroethene, tetrachloroethene, trichloroethene, vinyl chloride, and benzene, toluene, ethylbenzene, and xylene (BTEX).[169]

The zone of potential groundwater discharge for Philip Services/Burlington Environmental was estimated to be the area between RM 1.1 and RM 1.8 on the east side of the LDW, based on groundwater flow information (Appendix G in Windward 2003a); however, the groundwater discharge zone for this facility has not been formally delineated. None of the groundwater COCs that have SMS criteria were detected above the SQS in surface sediment samples collected from the zone of potential groundwater discharge (see Map 4-14b). None of the VOCs were detected in the three surface sediment samples analyzed from the zone of potential groundwater discharge.

Two seep samples have been collected from the zone of potential groundwater discharge (Windward 2004b; Weston 1999a) (Map 9-17). With the exception of barium and manganese, all of the groundwater COCs were analyzed in the seep samples. Cadmium, copper, and lead were detected in both seep samples, and nickel and silver were detected in one seep sample each.

One porewater sample was collected from within the zone of potential groundwater discharge; this sample was analyzed for metals and organometals. Of the metal groundwater COCs that were analyzed, only barium, lead, and manganese were detected in porewater.

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[169] Chemicals were selected as groundwater COCs for Philip Services/Burlington Environmental if they were retained after the first step in the aquatic ERA conducted by Philip Services (PSC 2001b) and they were detected in monitoring wells closest to the LDW.

In 2004, a subsurface barrier wall and a groundwater recovery system were installed. As of 2006, this system was still being monitored (Ecology and Environment 2008b).

Fate and transport analyses conducted as part of the FS for the Philip Services/ Burlington Environmental facility (Geomatrix 2006) indicated that COCs, including chlorinated ethenes, 1,4-dioxane, and possibly vinyl chloride, could reach the LDW at concentrations greater than the surface water protection criteria.[170] Ecology is still assessing groundwater at this facility as it progresses toward a new RCRA permit and Agreed Orders between Ecology and the Philip Services Corporation and as part of their source control program for the LDW.

Former Rhône-Poulenc Facility

The former Rhône-Poulenc facility is located on the east side of the LDW between RM 4.0 and RM 4.2 (Map 9-17). Multiple environmental investigations have been conducted at the site under RCRA since the 1990s (see Appendix I, Section I.4.7). Groundwater data evaluated in the groundwater pathway assessment were obtained from three separate reports: an RFI report (Rhône-Poulenc 1995), a sewer sediment technical document (Rhône-Poulenc 1996), and a groundwater monitoring report (GeoEngineers 2002). As of 2007, over 10 years of groundwater monitoring has been conducted at the site (Ecology and Environment 2008b). Groundwater COCs that were selected for evaluation in the groundwater pathway assessment were arsenic, chromium, copper, lead, mercury, vanadium, zinc, and BTEX.[171]

The zone of potential groundwater discharge for the former Rhône-Poulenc facility was estimated in the groundwater pathway assessment (Appendix G in Windward 2003a) to be the sediment area adjacent to the site (RM 4.0 to RM 4.2 on the east side of the LDW) based on the proximity of the facility to the zone and groundwater flow information; however, the groundwater discharge zone for this facility has not been formally delineated. Of the chemicals with SMS criteria that were identified as groundwater COCs for this facility, only mercury has been detected above the SQS (i.e., one CSL exceedance) in surface sediment samples collected from the zone of

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[170] The final RI cleanup levels selected for the water table, shallow, and intermediate groundwater were the minimum concentrations based on the following: calculated MTCA Method B groundwater cleanup levels based on an Asian and Pacific Islander exposure scenario for the consumption of fish for the groundwater-to-surface water exposure pathway; AWQC based on only the human health consumption of organisms (Section 304 of the federal CWA); ERA surface water screening levels protective of aquatic biota in surface water; AWQC freshwater and marine criteria maximum concentration, criteria continuous concentration, and organoleptic effects (Section 304 of the federal CWA); Washington State freshwater and marine acute and chronic effects criteria; and MTCA Method A cleanup levels (Geomatrix 2006).

[171] The groundwater COCs identified in the groundwater pathway assessment for the former Rhône-Poulenc facility were chemicals that were detected above action levels in groundwater (i.e., the lowest of the federal or state surface water quality standards for either fresh or marine water) in the monitoring wells closest to the LDW.

potential groundwater discharge (Map 4-14e). One surface sediment sample collected within this zone was analyzed for VOCs; BTEX was not detected in the sample (Appendix G in Windward 2003a).

Several seep samples were collected from the zone of potential groundwater discharge for Rhône-Poulenc as part of an RFI (Rhône-Poulenc 1996) (Map 9-17). All of the groundwater COCs identified in the groundwater pathway assessment were analyzed in the seep samples and were detected in at least one sample.

Porewater samples have also been collected from the zone of potential groundwater discharge as part of the LDW site inspection (Weston 1999a) and the Rhône-Poulenc sediment and porewater investigation (EPA 2005d) (Map 9-17). All of the metals identified as groundwater COCs were analyzed in the porewater sample collected as part of the LDW site inspection; arsenic and lead were the only COCs detected (Weston 1999a). All of the chemicals identified as groundwater COCs were analyzed in the porewater samples collected as part of the Rhône-Poulenc sediment and porewater investigation (EPA 2005d). Copper, mercury, and zinc were detected. Arsenic, chromium, lead, vanadium, and BTEX were not detected.

EPA's LDW site inspection (Weston 1999a) used sediment centrifugation to collect porewater samples. The Rhône-Poulenc investigation (EPA 2005d) used a mini-piezometer and seepage meter to collect porewater samples. Barium, iron, and manganese were the only metals that were detected in samples from both studies. The concentrations of barium and iron detected in the LDW site inspection porewater samples were within the range of the barium and iron concentrations detected in the Rhône-Poulenc investigation porewater samples. However, the concentration of manganese detected in the LDW site inspection sample was nearly three times higher than the maximum manganese concentration detected in the Rhône-Poulenc investigation porewater samples.

Recent groundwater monitoring and geoprobe sampling in an area along the west-central shoreline detected toluene, arsenic, and copper (Ecology and Environment 2008b). Recent groundwater data are summarized in Appendix I (Section I.4.7). Arsenic and copper concentrations in the sediment samples collected from the southwest shoreline area were less than the SQS. EPA is currently investigating the toluene groundwater contamination in the southwest corner of the East Parcel (Ecology and Environment 2008b), in accordance with the revised East Parcel corrective measures implementation work plan (Geomatrix 2007).

South Park Landfill

The South Park Landfill is located approximately 2,000 ft to the southwest of the LDW at RM 2.6 (Map 9-17). The facility has been a MTCA site since the late 1980s, and an independent remedial investigation and quarterly groundwater monitoring program were implemented at the facility in 1997 (Appendix G in Windward 2003a). Groundwater data collected in the late 1990s and early 2000s were evaluated in the groundwater pathway assessment. Vinyl chloride was the only groundwater COC identified in downgradient wells at the facility when the groundwater pathway assessment was conducted; therefore, it was the only chemical selected for evaluation in the assessment.

The zone of potential groundwater discharge for the South Park Landfill was not estimated in the groundwater pathway assessment, but could be between approximately RM 2.6 and RM 3.0, based on the fact that groundwater flows to the northeast from this site (Appendix G in Windward 2003a); however, the groundwater discharge zone for this facility has not been formally delineated. One surface sediment sample collected between RM 2.6 and RM 3.0 on the west side of the LDW has been analyzed for vinyl chloride; vinyl chloride was not detected in the sample. One seep sample was collected from the west side of the LDW near RM 3.0 as part of the RI (Windward 2004a). Vinyl chloride was analyzed in this sample and was not detected. One porewater sample was collected within the navigation channel at approximately RM 2.65; the sample was not analyzed for vinyl chloride.

Long-term groundwater monitoring is ongoing at the South Park Landfill. During the past two semi-annual monitoring events, conducted in October 2008 and March 2009, vinyl chloride concentrations in groundwater at the downgradient monitoring wells were non-detect at the laboratory practical quantitation limit of 0.2 µg/L (Goldberg 2009). Ecology is still assessing groundwater at this facility as part of an Agreed Order signed with the City of Seattle and the current property owner (South Park Property Development, LLC) and as part of their source control program for the LDW (Ecology 2009). Preparations are underway for initiation of an RI/FS at the South Park Landfill (see Section 9.3.2.5 for additional information).

T-108 (former Chiyoda Property)

The T-108/former Chiyoda property is located on the east side of the LDW between RM 0.5 and RM 0.7 (Map 9-17). Numerous environmental investigations have been conducted at the facility (see Appendix I, Section I.4.1). Groundwater data evaluated in the groundwater pathway assessment were collected on the eastern portion of the facility in 1992 as part of a site assessment summary (AGI 1992). Chemicals that were considered to be groundwater COCs in the groundwater pathway assessment were arsenic, cadmium, chromium, copper, lead, nickel, zinc, total HPAHs, total LPAHs, and total PCBs.[172]

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[172] Chemicals selected as groundwater COCs for T-108 in the Phase I groundwater pathway assessment (Appendix G in Windward 2003a) were those detected above MTCA Method A or C levels in any of the onsite wells that were also detected in wells closest to the LDW in 1992 (the most recent groundwater data available at the time of the assessment). In addition, copper, nickel, and zinc were selected because they exceeded federal AWQC, and PCBs were included as a conservative measure (Appendix G in Windward 2003a).

The zone of potential groundwater discharge for T-108 was estimated to be the sediment area adjacent to the site, from RM 0.4 to RM 0.7, based on the proximity of the facility and groundwater flow information (Appendix G in Windward 2003a); however, the groundwater discharge zone for this facility has not been formally delineated. In general, groundwater at the site flows radially in all directions from a relative high near the center of the facility. This includes some relative flow to the north toward the Diagonal Avenue S CSO/SD system; the potential interaction of groundwater with this drainage network is unknown.

Of the groundwater COCs, cadmium has been detected above the SQS in one surface sediment sample collected within the zone of potential groundwater discharge, total HPAHs have been detected above the SQS in two samples, and total PCBs have been detected above the CSL in two samples and above the SQS in several samples (see Map 4-14a). No seep or porewater samples have been collected from the zone of potential groundwater discharge.

The shallow groundwater samples from the T-108 site were collected primarily within dredge-fill material, above the native tideflat surface. PCBs were detected at low concentrations in samples collected before 1991, but not in any consistent pattern. PCBs were not detected in samples collected in 1991 and 1992. The 1991/1992 data did not detect PAHs above MTCA Method C levels.

Arsenic, cadmium, chromium, lead, and zinc were detected in a few monitoring wells at concentrations above the MTCA Method C cleanup criteria. Given the fine-grained nature of the onsite soils, and the old high-flow groundwater sampling techniques used, it is suspected that some of the metals detected in the groundwater samples may be associated with fine soil particles suspended in the water samples and subsequently dissolved in the acid extraction conducted for laboratory analysis (Appendix G in Windward 2003a). Of these metals, none exceeded the SQS in the surface sediment in the zone of potential groundwater discharge.

Elevated concentrations of PCBs have been detected in sediments adjacent to the site; however, based on the groundwater data for the site and the known high attenuation of PCBs in groundwater, it is likely that a source other than groundwater migration was responsible for elevated PCBs in LDW sediment adjacent to this site (Appendix G in Windward 2003a).

An additional groundwater monitoring program was conducted by the Port in 2006 and 2007 to further evaluate the T-108 groundwater pathway to LDW sediment (Pacific Groundwater Group 2006, 2007). Groundwater data are summarized in Appendix I (Section I.4.1). Chemicals detected in groundwater in the four rounds of sampling conducted as part of this program were arsenic (total and dissolved), chromium (total and dissolved), copper (total and dissolved), lead (total only), zinc (total and dissolved), and four individual PAHs.[173] Arsenic (total and dissolved) and lead (total only) were the only chemicals detected above MTCA Method A cleanup levels for groundwater. These exceedances were detected in samples collected during the first two rounds of sampling; arsenic and lead concentrations were below MTCA Method A cleanup levels in the third and fourth sampling rounds. The report concluded that the groundwater pathway for T-108 should be considered closed and that groundwater monitoring should be discontinued (Pacific Groundwater Group 2007). Upon review of the associated documentation, Ecology concurred with this recommendation and acknowledged that groundwater at T-108 was not a potential source of contamination to adjacent sediment in the LDW.

9.4.7 Bank erosion

Erosion of contaminated bank soils and sediments can be an ongoing source of contamination to the LDW, as discussed in Section 9.1.6. Unprotected shoreline banks are potentially susceptible to erosion by both wave action and surface water runoff. Most of the shoreline area along the LDW has been developed over the past 100 years for industrial, commercial, or manufacturing purposes, and currently, the majority of the LDW shoreline is armored with constructed steel, wood, concrete bulkheads, sheet pile walls, and riprap revetments. While the presence of shoreline armoring serves to reduce bank erosion in many areas, it does not eliminate the potential for contaminated bank material to reach LDW sediments. Tidal exchange, groundwater discharge, and soil and sediment erosion can occur even in areas that are armored, particularly when armoring is patchy or in disrepair. These processes have the potential to carry chemicals from bank areas to LDW sediments.

Characterization of erodible material is required at sites where bank construction or remediation is planned. Such characterization has been completed at facilities within several of the SCAs in the LDW. At the Slip 4 and T-117 SCAs, PCB contamination has been found in bank soils (see Appendix I). At both of these sites, the erosion of bank material was identified as a direct source of contamination in sediment (Ecology 2005a, 2006a). Remediation activities at these two EAAs, both planned and ongoing, will include removal or capping of contaminated bank soils to eliminate this pathway as a potential source of sediment recontamination.

In addition to bank characterization at the Slip 4 and T-117 SCAs, bank sampling has also been conducted at several other shoreline facilities along the LDW, including Boeing Plant 2, Jorgensen Forge, T-108, and the Industrial Container Services/Trotsky property. Bank soil data for these facilities are presented in Appendix I. Bank soils at the Rhône-Poulenc and Douglas Management Company facilities have also been sampled; however, the data were not available for inclusion in Appendix I (Thomas

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[173] The results for one well in the second round of sampling were R-qualified (rejected). These results are not included in this summary or in Appendix I.

2009). Bank soil sampling has also been conducted at several of the restoration sites along the LDW, including Herrings House, Hamm Creek, and the Upper Turning Basin (Cargill 2009).

9.4.8 Routine waterway activities

Potential sources of contamination from routine activities on the LDW include dredging or in-water structure maintenance, which may resuspend contaminated material. In addition, chemicals may leach from structures such as piers or pilings, or originate from recreational and commercial boats or overwater cargo transfer. Vessel movement can potentially resuspend contaminated sediment by a process known as propeller wash. Further discussion of these activities is presented below.

9.4.8.1 Sediment resuspension activities

Dredging activities in the LDW have the potential to resuspend contaminated sediments. In-water construction projects, including the removal of old structures, also have the potential to resuspend sediments. To minimize sediment resuspension during dredging, demolition, or construction activities, appropriate BMPs are required. As part of the design phase of the project, BMPs, compliance protocols, and other requirements are identified and monitored. The most common types of BMPs are operational controls or the use of special equipment.

Operational controls are used to minimize the resuspension of sediment. These controls can include slowing down operations, making sure that barges are not overfilled, and containing or filtering leaks and/or spills from barges. Careful sequencing of the dredging operations by site, area, or lift can greatly minimize potential impacts to the waterway from dredging activities. The use of special equipment can also minimize the resuspension of sediment. Special equipment can include specially constructed dredging buckets designed to reduce increased turbidity of suspended solids (sometimes referred to as environmental or closed-bucket dredging), and silt curtains or Gunder booms designed to limit the distribution of suspended sediment. Water quality monitoring is also typically conducted during dredging and construction activities to ensure that potential water quality impacts are minimized.

Propwash, which generally occurs in or near berthing areas or in shallow areas LDW, may also resuspend sediment. The effect of propwash can vary depending on the sediment type, water depth, and vessel type (see Section 3). Higher bed scours may occur in the bench areas than in the navigation channel from the effects of ships maneuvering in the shallower near shore areas (Windward and QEA 2008).

9.4.8.2 Other activities

Normal day-to-day in-water operations and boat traffic in the LDW can also potentially introduce chemical contaminants to the LDW. These potential sources are generally localized. Some of these activities and potential sources are listed below.

- Direct discharge from boats such as gray, bilge, or ballast waters
- Fuel and other releases from spills and operation and maintenance of boats and vessels
- Spills and releases from loading/offloading and other practices associated with docks, wharves, and piers
- Wastes and other materials from marinas and small boatyards

Ballast is material placed on a vessel to control its stability and buoyancy. In general, ballast is comprised of water, sediment, or a combination of water and sediment. Regulations on ballast water from vessels 300 gross tons or greater in size are implemented by WDFW under RCW Chapter 77.120 (WDFW 2008). These regulations prohibit the discharge of ballast water to waters of the State of Washington, except in cases of open sea exchange, when the ballast water has been treated, or when it consists entirely of "common waters" ballast. Common waters ballast is that taken from designated areas off the coasts of Washington, Oregon, and British Columbia, which are specified by WDFW. The USCG has also implemented a national ballast water management program to enforce the regulations of 33 CFR 151, Subparts C and D. These regulations apply to all vessels with ballast tanks, regardless of size. The program implements mandatory ballast water management practices including regular cleaning and removal of sediments from ballast tanks, practices to limit ballast water discharge, and requirements for vessel-specific ballast water management plans (USCG 2008). Liquid wastes, such as bilge water (water that accumulates in the spaces between lower compartments of a boat and its hull), originate from marinas and boatyards and have the potential to contaminate the LDW. These wastes are regulated under Ecology's NPDES program. Bilge water can contain various chemicals, including oil and detergents (Port of Seattle 2007a).

Boatyards and marinas that conduct maintenance activities out-of-doors are required to maintain a general boatyard NPDES permit, or potentially an industrial NPDES permit. The boatyard general permit contains mandatory BMPs that must be followed for the management of bilge water. These regulations prohibit discharge of bilge waters to the waters of the state if they contain solvents, detergents, or other chemicals, or if they cause any visible sheen when discharged. Gray water must be discharged to the sanitary sewer.

Spill response and tracking programs have been established by the USCG, the Ecology, and SPU. These programs have established response and cleanup methods for both in-water and upland spills. These programs help provide spill source control for the LDW by containing and cleaning up reported spills that occur directly to the water or within drainage systems that discharge to the waterway. Tables G-14 and G-15 in Appendix G list spills reported in the LDW and the LDW drainage basin.

9.5 SUMMARY

The LDW is a dynamic system that has been altered by over 100 yrs of industrial, commercial, and residential development. Potential sources of contamination to LDW sediment are numerous and may include both historical and ongoing operations and various pathways. Data gaps associated with the characterization of potential chemical sources to the LDW still exist; source control investigations started in the late 1980s and are ongoing.

This section is a summary of source control efforts and investigations with information available up to the date of the draft final version of this document (July 17, 2009). The cutoff date for inclusion of numerical data in this section was a collection date of December 31, 2007, and a reporting date of July 11, 2008, based on discussions with EPA and Ecology. Work to fill source control data gaps is expected to continue for years to come. Ecology will make site-specific recommendations on the status of source control and the potential for recontamination as source control work continues.

Earlier sediment investigations led to the identification of seven candidate EAAs for further sediment characterization and/or remediation; removal actions have already been implemented at two of these seven locations and removal actions are currently being planned for three more EAAs.

Ecology is leading the source control evaluation and assessment of the 23 SCAs, which includes the seven EAAs. Ecology has or will prepare SCAPs and data gaps reports to help direct future investigation and remediation efforts within these areas while assessing potential sources for the chemicals identified.

Modern regulatory requirements have established inspection and compliance programs to monitor and control the handling and disposal of manufacturing wastes produced by industrial operations along the LDW. These regulations have helped reduce sources of contamination that were common practices in the past.

Air emissions are a potential ongoing source of many chemicals to the LDW. Two hundred businesses in the greater LDW basin are registered as active sources of air pollutants. Programs are in place to help reduce these emissions over the long term. However, motor vehicles and the burning of wood and fossil fuels will remain ongoing sources of chemicals to the atmosphere until technologies evolve and these sources can be controlled.

Permit programs are in place to help monitor and control direct discharges to the LDW from CSOs, EOFs, and SDs in the basin. Since 1970, the City of Seattle has reduced the annual CSO volume citywide by approximately 95%; and since 1988, King County has reduced its annual CSO volume by 63%. Direct discharges to the LDW contribute approximately 1% of the annual sediment load to the LDW compared with inputs from the Duwamish/Green River system. Note that this 1% refers to sediment loading and not chemical loading; direct discharges to the LDW may have a local impact at the point of discharge. Although CSO control programs and stormwater and wastewater discharge permits play an important role in source control, they have not been developed specifically to protect sediment quality.

Source sampling in the LDW study area has been conducted by several parties, including the City of Seattle, King County, the Port of Seattle, Ecology, and private companies. Most of the sampling conducted to date has focused on tracking potential sources so that source controls can be put in place to reduce the amount of contaminants discharged to the LDW. Various sampling techniques have been used to identify and characterize contaminant sources, including source sediment sampling (e.g., catch basin, inline sediment grabs, and inline sediment traps), as well as stormwater and wastewater sampling. These samples have been collected as part of business inspection programs, environmental investigations at individual facilities, and source-tracing investigations (see Section 9.4.4.7 for more information). Results from January 2002 through December 2007 indicate that zinc, BEHP, and BBP frequently exceeded the CSL in solids samples collected as part of source-tracing investigations.

Ongoing monitoring of industrial and municipal stormwater discharges to the waterway will provide useful information to help determine which chemicals might be migrating to sediments in the LDW while simultaneously helping to evaluate the success of any source control measures taken upland of the monitored locations. Monitoring requirements will be established in the ROD.

Chemicals identified in upland groundwater or seeps along the LDW have generally not been shown to accumulate in adjacent sediments, except in a few small areas (e.g., the head of Slip 1). Most chemicals identified in upland groundwater samples are likely to be diluted or attenuated prior to discharge.

Although the majority of the river bank along the LDW is covered or armored with impervious materials, erosion of contaminated bank material into the LDW is still possible. This potential source of chemicals to the waterway is currently not well characterized.

10 Key Findings

This section highlights the important findings of the RI and synthesizes these findings into a unified CSM. The physical, chemical, and ecological processes that have been examined in the RI are interrelated and must be considered in combination when assembling and analyzing alternatives in the FS, and when making remedy and risk management decisions for the LDW.

10.1 GEOGRAPHIC LANDSCAPE OF THE LDW

Prior to the significant alterations that began in the early 1900s, the lower sections of the Duwamish River were characteristic of the many estuarine rivers in the Pacific Northwest. The river exhibited high sinuosity as it meandered through the valley floor and emptied into Elliott Bay as a broad delta. Given the nature of the rivers feeding the lower Duwamish and the flat topography of the valley floor, the terminal section of the river was highly depositional, giving rise to the broad expanse of mudflats depicted in early maps of the area.

As a result of its development, the Green River/Duwamish River system has undergone significant hydrological changes, which have reduced flow by an estimated 78% from historical levels. These changes have included the natural diversion of the White River following the flood of 1906, the loss of flow from the Black River when the water level in Lake Washington dropped in response to the opening of the Ship Canal and Hiram M. Chittenden locks, and the diversion of the Cedar River to Lake Washington. Even with the loss of these water sources, the Green River/Duwamish River system continued to flood low-lying areas. The Howard Hansen Dam, built in 1961 for flood control, effectively decreased peak flows in the river system. Currently, peak flows rarely exceed 12,000 cfs. Despite these changes, the Green River still delivers an estimated 221,000[174] metric tons of sediment load to the Duwamish River each year (QEA 2008).

A portion of the LDW is maintained as a federal navigation channel by USACE. Elevations within the navigation channel are maintained between -30 ft MLLW and -15 ft MLLW, with the shallower elevations maintained in the upper reaches of the LDW. King County Waterway Commercial Waterway District No. 1 constructed an Upper Turning Basin at the head of the navigation channel (RM 4.6) prior to 1920, which serves as a catchment for sediment from the Green/Duwamish River system. The need for this engineered catchment, along with the periodic dredging required to

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[174] Estimated sediment load from the Green River to the LDW from 1960 to 1980 (QEA 2008). This value remains relevant because no significant changes have occurred to the morphology of the river that would be expected to change the sediment load.

maintain the navigation channel, is evidence that the LDW continues to be a depositional environment.

The lower 5 mi of the Duwamish River have been highly modified. In the early 1900s, issues with flooding in the area precipitated the installation of levees and dams and the subsequent channelization of the river. Most of the tidal marsh, mudflats and shallows, and tidal wetlands historically present in the lower Duwamish River have either been filled or dredged. Today, the majority of the LDW shoreline consists of riprap, pier aprons, or sheet pilings, especially in the downstream portion of the site.

Intermittent, shallow benches exist in the nearshore, intertidal, and shallow subtidal zones of the LDW outside the navigation channel. These benches are of various dimensions and elevations, with minimum elevations of less than 3 ft MLLW. Intertidal habitats are dispersed in relatively small patches (i.e., generally less than 1 ac in size) downstream of RM 3.0, with the exception of Kellogg Island, which represents the largest contiguous area of intertidal habitat remaining in the LDW. Low intertidal mudflats are present below upper bank riprap and miscellaneous armoring in the reaches upstream of RM 3.0.

The LDW provides a navigational corridor for the movement of raw materials and products used by local facilities, and serves as a major base for marine transport to Alaska. Current land use, zoning requirements, and land ownership within most of the LDW corridor are consistent with the characteristics of an active industrial waterway. Non-industrial uses also occur on and along the waterway. Two mixed residential/commercial/industrial neighborhoods, South Park and Georgetown, are located west and east, respectively, of the LDW. The presence of these neighborhoods and various public access points along the LDW is associated with recreational use of the waterway. Recreational users can collect seafood from the various public access areas (although there is a health advisory in effect advising against consumption of resident fish and shellfish), can walk or play on the beaches, or can access the river by boat (most of the LDW is publicly owned and accessible by boat). In addition, the LDW is also one of the locations of the Muckleshoot Tribe’s commercial, ceremonial, and subsistence fishery for salmon. The Suquamish Tribe actively manages aquatic resources north of the Spokane Street Bridge, located just north of the LDW. The Duwamish Tribe uses Herring’s House Park and other parks along the Duwamish for cultural gatherings.

Despite the highly modified river habitat, the LDW is home to a diverse assemblage of aquatic and wildlife species and a robust food web that includes top predators. More than 53 resident and anadromous fish species are found in the LDW (Windward 2004d, 2005b, 2006c). A total of 61 invertebrate taxa were identified in a survey of the intertidal and subtidal benthic invertebrate community, including crabs, clams, mussels, crustaceans, marine worms, and gastropod species (Windward 2005e). In addition, surveys have indicated that up to 87 species of birds and 6 species of mammals use the LDW during part of the year to feed, rest, or reproduce. Avian species include passerines, raptors, shorebirds, waterfowl, and seabirds (1979); mammals include river otters, which may be permanent residents, and harbor seals and sea lions, which may visit the area. Fifteen species identified in the LDW study area are listed under the ESA or by WDFW as candidate species, threatened species, or species of concern.

While the LDW will continue to be used as an industrial corridor, there is a high interest by the resource trustees and the public to create higher quality habitat where opportunities exist. The focus of these efforts thus far has been in small areas of marsh and unvegetated intertidal marsh habitat, including T-105 (RM 0.1); T-107 and Herring's House Park near Kellogg Island (RM 0.3 to RM 0.5); T-108 (RM 0.6); GSA marsh restoration (RM 0.8); Hamm Creek (RM 4.3); Muckleshoot Kenco Marine, Coastal America, and Port of Seattle mitigation sites near the Upper Turning Basin (RM 4.7); and Cecil B. Moses Park (North Winds Weir, RM 5.7).

10.2 CONCEPTUAL SITE MODEL

A CSM for the LDW that describes key features and processes of the waterway was developed as part of the RI/FS process. The CSM will aid in the assembly and evaluation of remedial action alternatives in the FS and in future decision-making for the site. The CSM is considered a working model throughout the RI/FS process, and may continue to be refined as new information and analyses are developed.

The LDW CSM provides a synthesis of the major processes governing the movement and distribution of chemicals at the site as well as potential exposure pathways for the people and animals that use the site (Figure 10-1). Three key components of the LDW CSM are worthy of discussion because of their importance:

- **Physical CSM –** The influence of hydrodynamic and sediment transport processes on the physical distribution of sediment within the LDW
- **Chemical CSM –** Patterns in the distribution of chemicals in the LDW
- **Exposure CSM –** Exposure of human and ecological receptors to chemicals within the LDW

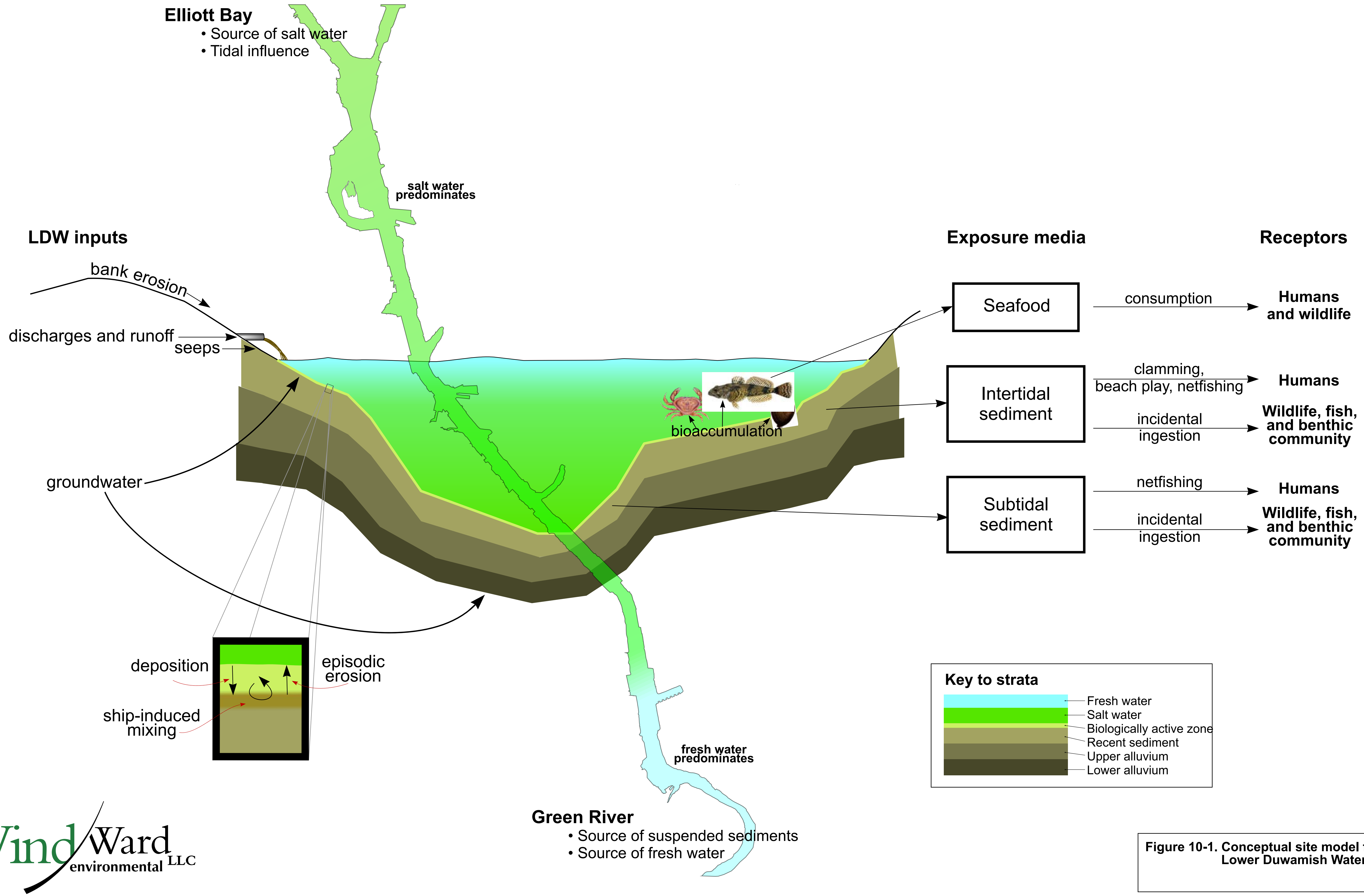

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Figure 10-1. Conceptual site model for the Lower Duwamish Waterway

10.2.1 Physical CSM

The LDW can be characterized as a well-stratified salt wedge estuary. The LDW receives inflows of marine (or salt) water from Elliott Bay and fresh water from the Green/Duwamish River system. Fresh water from the Green/Duwamish River system moving downstream overlies the tidally influenced salt water entering from Elliott Bay. Typical of tidally influenced estuaries, the LDW has a relatively sharp interface between the fresher water outflow at the surface and saltwater inflow at depth.

During average-flow conditions in the Green River, the saltwater wedge extends, on average, up to approximately RM 4.5 to RM 4.8, depending on the tidal cycle and river flows (Figure 10-1). During high-flow conditions, the salt wedge may not extend upstream of Slip 4 (RM 2.9). Under extreme conditions, such as very high tide stages during periods of low freshwater inflow, the saltwater wedge can extend as far upstream as RM 8.7. Thus, much of the LDW is dominated by two-layer estuarine circulation, in which the denser salt water is transported upstream in the lower layer of the water column and the fresh water is transported downstream in the upper layer. Above RM 4.8, the LDW is a tidally influenced river in which the entire water column is generally composed of fresh water.

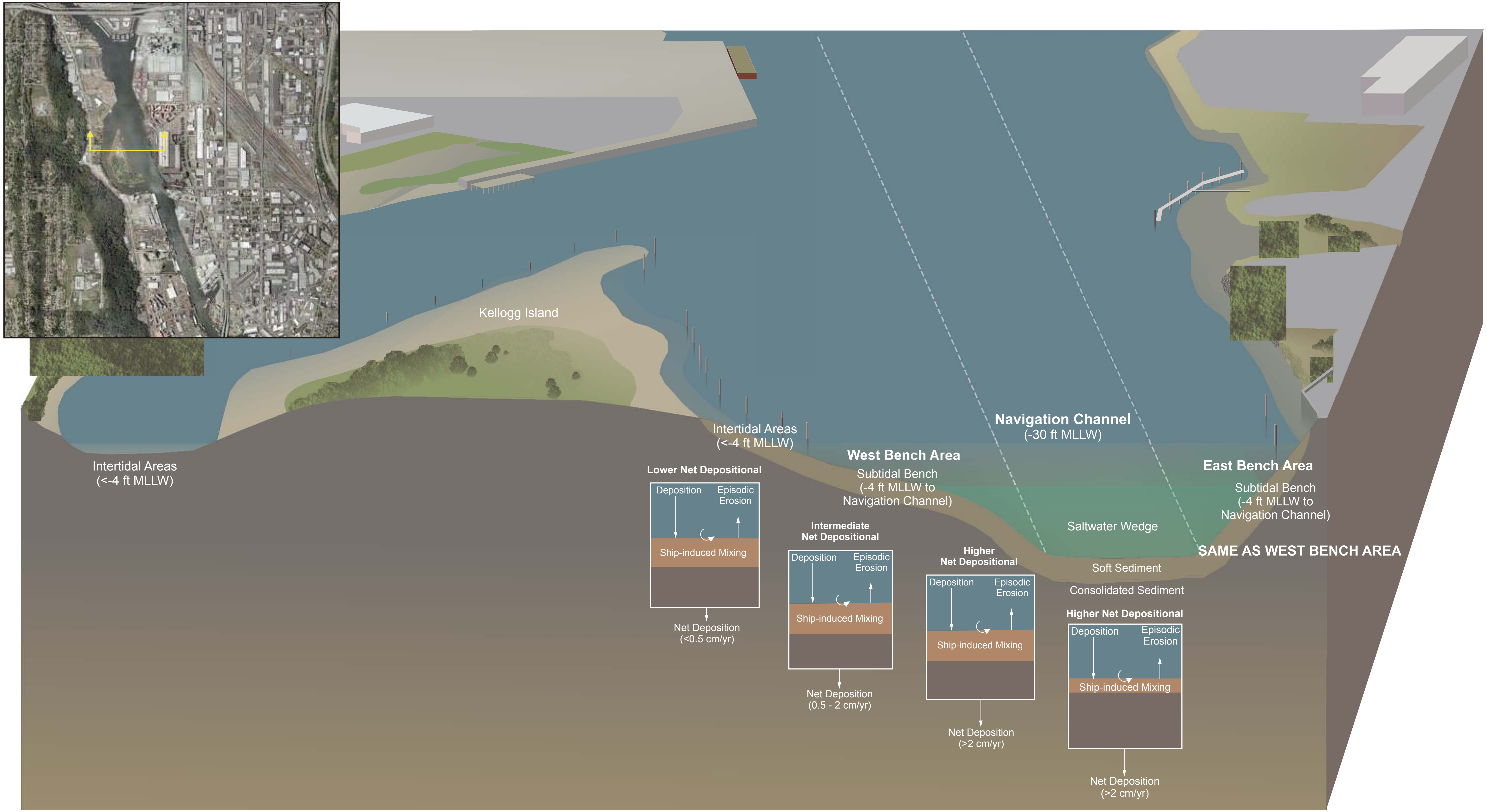
The two-layer estuarine circulation plays a key role in patterns of sediment deposition and scour potential. Reaches of the LDW that are occupied by the saltwater wedge experience minimal bed scour from fresh water coming from upstream because much of the sediment bed is covered by tidally driven salt water. The tidal velocities within the saltwater wedge are much lower than the current (river) velocities. Reaches of the LDW that are entirely fresh water during high-flow events are more susceptible to bed scour because the near-bed velocities are driven by river flow and not tidal velocities.

The physical CSM, which is based on both empirical data collected from the LDW and an STM, identified three reaches of the LDW that have distinct sediment transport characteristics: RM 0.0 to RM 2.2 (Reach 1), RM 2.2 to RM 4.0 (Reach 2), and RM 4.0 to RM 4.8 (Reach 3). Differences among these reaches are defined by the hydrodynamic characteristics of each reach, particularly during high-flow events. The descriptions of reaches represent general conditions within each reach; conditions at specific locations within each reach may vary because of site-specific influences (e.g., berthing, outfalls, dredging activities). The sediment dynamics for Reaches 1, 2, and 3 are illustrated in Figures 10-2, 10-3, and 10-4, respectively, and summarized below.

- **Reach 1: RM 0.0 to RM 2.2 —** The permanent presence of the saltwater wedge in this reach serves as a protective barrier for the river bed. Consequently, the potential for bed scour, even during high-flow events, is relatively low, as is the potential for buried sediments to be re-exposed. This reach is net depositional on an annual time scale. Approximately 60% of the area within and outside the navigation channel has net sedimentation rates between 1 and 2 cm/yr; an additional 30% of the area has net sedimentation rates greater than 2 cm/yr.

- **Reach 2: RM 2.2 to RM 4.0 —** Reach 2 is occupied by the saltwater wedge during most flow conditions, but during high-flow events the toe of the saltwater wedge may only reach the downstream portion of this reach. As a result, Reach 2 could experience net erosion during high-flow events with 2-year or greater return intervals. Even during 100-year high-flow events, net erosion is generally limited to the upper 10 cm (i.e., within the biologically active zone; Section 2.8.2.2) (Ecology 2007n) of the sediment bed. Reach 2 is also net depositional on an annual time scale. Net sedimentation rates are estimated to be spatially variable. Between RM 2.2 and RM 2.6, approximately 94% of the area outside the navigation channel and all of the area within the navigation channel has net sedimentation rates greater than 2 cm/yr. Between RM 2.6 and RM 4.0, approximately 50% of the area both within and outside of the navigation channel has net sedimentation rates greater than 2 cm/yr.
- **Reach 3: RM 4.0 to RM 4.8 —** During most flow conditions, the saltwater wedge extends into this reach, with the location of the toe of the wedge dependent on a combination of river-flow and tidal influences. During high-flow events, the saltwater wedge is not present in this reach, and freshwater flows dominate. This reach is wider than the downstream reaches, which creates lower current velocities that are less likely to erode the sediment bed. The upstream portion of the reach serves as a catchment for sediments transported into the LDW from upstream; by dredging this area every several years, the USACE is able to keep a large portion of the sediments (especially bedload) entering the LDW from upstream from moving into the lower reaches of the LDW.

Over the entire LDW, approximately 50% of the area both inside and outside of the navigation channel has long-term (i.e., 30 years) net sedimentation rates greater than 2 cm/yr. Over shorter time scales (i.e., days to weeks) associated with high-flow events, approximately 80% of the LDW area is expected to be net depositional. Slightly more erosion is estimated for the 100-year high-flow event (12,000 cfs) compared with the 10-year (10,800 cfs) and 2-year (8,400 cfs) high-flow events, but the areas experiencing erosion as well as the net erosion depths within those areas are similar for all high-flow events (Maps 3-4, 3-5, and 3-6). Approximately 6% of the LDW is estimated to experience greater than 10 cm net erosion during a 100-year high-flow event, with a maximum estimated net erosion depth of approximately 21 cm. Based on these model results, high-flow events are not expected to expose subsurface sediment at greater depths.



Notes: 1. Approximate net depositional rates from Sediment Transport Analysis Report, Windward and QEA 2006.
2. Inserts are qualitative illustrations and are not to scale.

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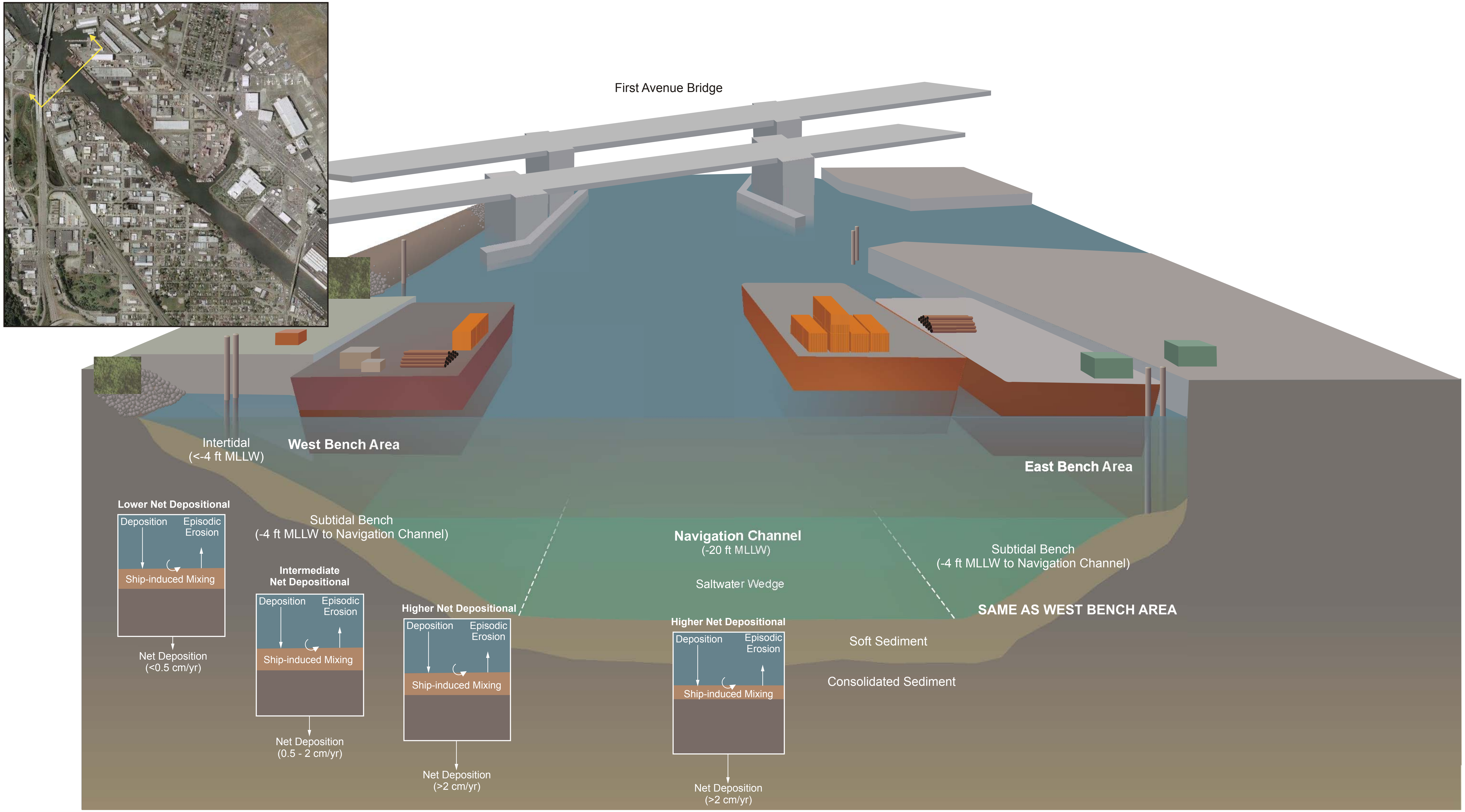
NOTE: Figure prepared January 11, 2006, by Brick Tudor Studios for RETEC, Inc. (RETEC | ENSR).
Updated July 25, 2008 by ZAC, Windward Environmental LLC

Figure 10-2. LDW sediment transport CSM for Reach 1 (RM 0.0 to RM 2.2)



Notes: 1. Approximate net depositional rates from Sediment Transport Analysis Report, Windward and QEA 2006.
2. Inserts are qualitative illustrations and are not to scale.

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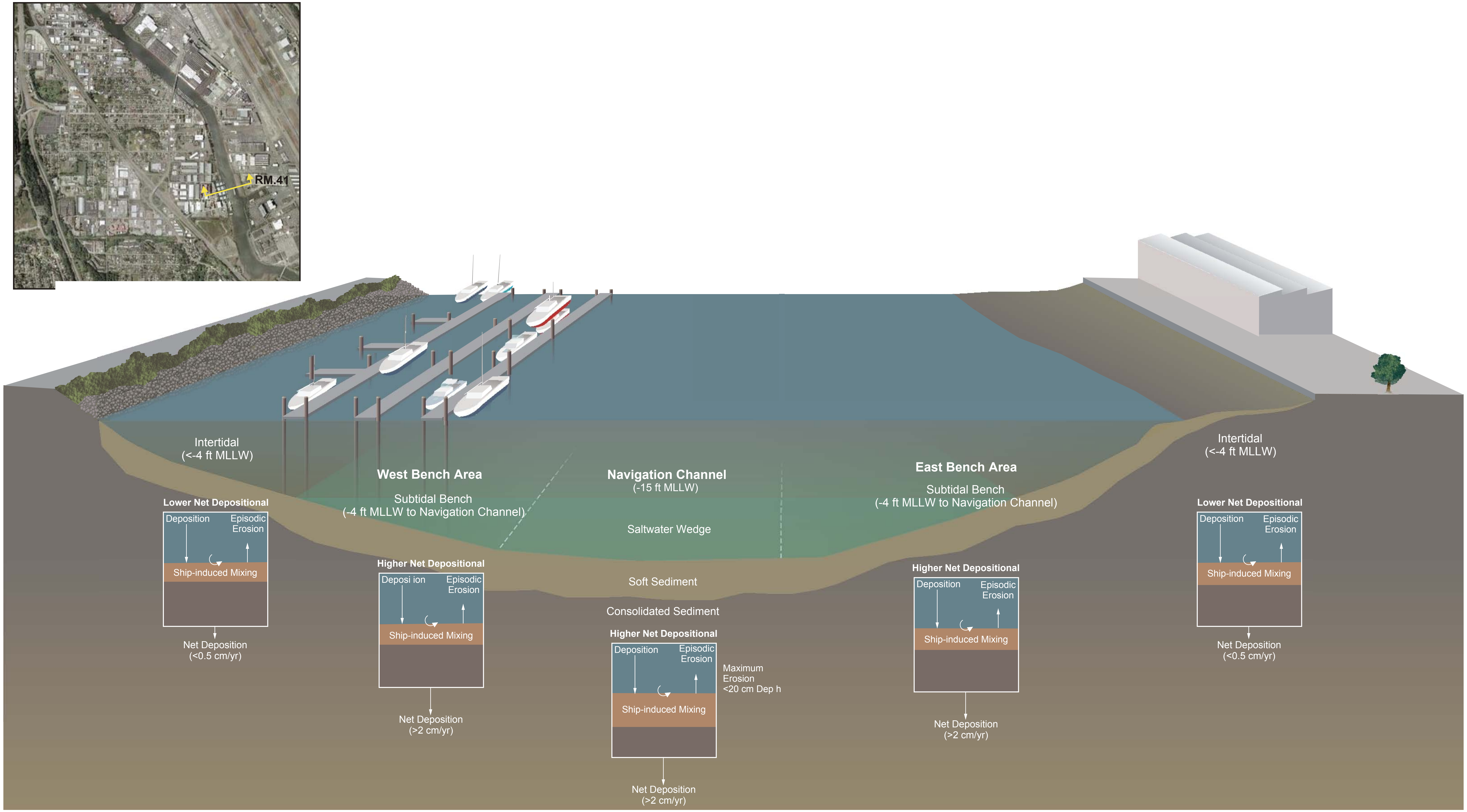
NOTE: Figure prepared January 11, 2006, by Brick Tudor Studios for RETEC, Inc. (RETEC | ENSR).
Updated July 25, 2008 by ZAC, Windward Environmental LLC

Figure 10-3. LDW sediment transport CSM for Reach 2 (RM 2.2 to RM 4.0)



Notes: 1. Approximate net depositional rates from Sediment Transport Analysis Report, Windward and QEA 2008.
2. Graphics and inserts are qualitative illustrations and are not to scale.

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NOTE: Figure revised August 2008 by Brick Tudor Studios for ENSR.

Figure 10-4. LDW sediment transport CSM for Reach 3 (RM 4.0 to RM 4.8)

On a smaller scale, there are a number of processes that can affect localized sediment deposition and erosion. Net sedimentation rates may be higher than estimated by the STM in the vicinity of specific outfalls if the actual particulate load from those outfalls is greater than assumed or not specifically included in the model. Net sedimentation rates in specific areas may also be higher than estimated by the model if there have been significant inputs of anthropogenic materials (e.g., sandblast grit in the vicinity of shipyards or bulk material spilled during offloading) that are not addressed by the STM. Prop wash associated with ship and tugboat operations in the LDW can also have localized effects and has the potential to erode deeper holes in small areas than the net erosion depths estimated by the STM. Ship-induced bed scour from passing vessels is estimated to result in a continual reworking of the uppermost 1 to 2 cm of the sediment bed within the navigation channel. In a typical event, sediment is suspended for a brief period of time, but is then re-deposited shortly thereafter once the propeller-driven forces are no longer present.

Based on a mass balance of sediment loads entering and leaving the LDW system, it is estimated that less than 1% of the sediment load enters the LDW from lateral inputs such as storm drains, CSOs, and surface water runoff. Over 99% of the sediment entering the system is from upstream. Approximately 50% of the incoming sediment exits the system as suspended sediment; the remaining material settles and becomes incorporated into the sediment bed. The sediment transported from upstream of the LDW is deposited onto the sediment bed and is mixed with the surface sediment layer (i.e., top 10 to 15 cm) by both biological and physical processes. Over time, the bed-source content of the surface layer decreases at an approximately exponential rate, primarily because of the deposition of upstream source sediment (Map 3-10). The STM report provides estimates of the rate at which sediment from upstream sources is added to bedded LDW sediment (QEA 2008). The sediment transport model developed for the LDW to estimate scour and deposition patterns was collaboratively developed and calibrated by a working group that included representatives from EPA, Ecology, USACE, and LDWG. Uncertainties in the STM are discussed in detail in the STM report (QEA 2008) and summarized in Section 3. The sediment transport working group (with representatives from EPA, USACE, and LDWG) considers the model to be functioning well for site-wide applications. At smaller spatial scales on the order of several grid cells (94% of the grid cells are 1 ac or less), the model results provide one line of evidence to guide decision-making.

10.2.2 Chemical CSM for sediment

The extensive RI dataset for both surface sediment (e.g., 0 to 15 cm in depth) and subsurface sediment provides a characterization of sediment contamination patterns within the LDW. The observed patterns in the spatial and vertical distributions of chemicals in surface and subsurface sediment, respectively, are the result of interactions among several factors, including the proximity and status of chemical sources (present-day and historical), mixing of the surface layer, transport and deposition of sediment within the LDW over time, and localized conditions that affect sediment mixing (e.g., scour from outfalls and tugboats, dredging).

Numerous chemicals are present in surface sediment in the LDW. Risk driver chemicals (referred to as indicator hazardous substances in the MTCA program [WAC 173-340-703]) were selected as part of the baseline risk assessments, in consultation with EPA and Ecology, to focus the remedial analyses in the FS.[175] The nature and extent of contamination are discussed in Section 4.

Four risk drivers were identified in the HHRA based on risks associated with seafood consumption and direct sediment contact: total PCBs, arsenic, cPAHs, and dioxins and furans. Total PCBs were also identified in the ERA as a risk driver for river otter. In addition, 41 chemicals (including total PCBs, arsenic, several phthalates, and various PAHs) were identified as risk drivers for benthic invertebrates because detected concentrations of these 41 chemicals exceeded the SQS of the Washington State SMS at one or more locations (Table 4-16). Total PCB concentrations in surface sediment exceeded the SQS at more locations than any other chemical. The chemical with the next highest number of locations with detected concentrations in surface sediment greater than the SQS was BEHP. The distributions of these risk driver chemicals in sediment are discussed below.

10.2.2.1 Surface sediment patterns

In general, high concentrations of chemicals were detected in localized areas separated by larger areas of the LDW with relatively low concentrations (Maps 4-70a through 4-70c). PCBs are widely distributed throughout the LDW and concentrations are highly variable. Total PCBs were detected in 94% of the 1,365 surface sediment samples in which they were analyzed, at concentrations ranging from 1.6 to 220,000 µg/kg dw, with a median concentration of 115 µg/kg dw (Maps 4-18 and 4-19). Arsenic was detected in surface sediments throughout the LDW, with concentrations ranging from 1.2 to 1,100 mg/kg dw, with a median concentration of 11 mg/kg dw (Maps 4-29 and 4-30). The highest arsenic concentrations were also highly localized.

Despite the widespread distribution of both of these chemicals in the LDW, the locations of the highest total PCB and arsenic concentrations were generally not in the same areas, indicating that sources likely differ for these two chemicals. The majority of the high total PCB concentrations were located within fairly well defined areas (Map 4-19), suggesting the nearby presence of one or more ongoing or historical sources. Areas with the highest total PCB concentrations in the baseline surface sediment dataset included the Duwamish/Diagonal EAA, EAA 2 (RM 2.2 on the west

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[175] Other chemicals that exceeded risk thresholds but were not selected as risk drivers for a particular exposure pathway will be addressed through a focused evaluation in the FS; these chemicals may also be re-assessed as part of the 5-year review or included in the post-remedial monitoring program, as appropriate.

shore), the Slip 4 EAA, the T-117 EAA, the Boeing Plant 2/Jorgensen Forge EAA, and the Norfolk EAA (Map 4-19).[176] Removal actions within some of the EAAs have already reduced surface sediment PCB concentrations in those areas. Likely sources of PCBs have been identified in most of these EAAs and source identification efforts are ongoing. However, specific sources have not been identified for all areas with higher total PCB concentrations (e.g., RM 1.0 in the navigation channel).

The majority of the higher arsenic concentrations were also located within fairly well-defined areas (Map 4-30), suggesting the presence of one or more nearby ongoing or historical sources. Areas with the highest arsenic concentrations in the baseline surface sediment dataset included the Duwamish/Diagonal EAA, Slip 1, RM 1.1 to RM 1.5, RM 2.0, RM 2.8, and RM 3.7 to RM 3.9 (Map 4-30).

Areas with the highest cPAH concentrations in the baseline surface sediment dataset were located in many of the same areas identified for arsenic and total PCBs, including the Duwamish/Diagonal EAA, RM 1.3 to RM 1.4, Slip 4, and RM 3.7 to RM 3.8 (Maps 4-37 and 4-38). Other areas with high cPAH concentrations were generally downstream of RM 1.1, at RM 2.1, RM 2.7, and in Slip 6. Sources of cPAHs for some of these areas have been identified. In general, high cPAH concentrations were more dispersed than PCBs and arsenic, suggesting more ubiquitous sources.

Except for a few areas with substantially higher dioxin and furan TEQs, dioxin and furan TEQs were generally uniformly distributed in the LDW (Section 7.3). Dioxin and furan TEQs were highest in the embayment on the west side of the LDW between RM 1.4 and RM 1.5 (Map 4-49). Other high concentrations were generally located in EAAs, such as the Duwamish/Diagonal EAA, EAA 2 at RM 2.2, and the Boeing Plant 2/Jorgensen Forge EAA. A few other locations also exist, such as the navigation channel at RM 1.0, where both the dioxin and furan TEQ and the total PCB concentrations were high.

Higher concentrations of numerous chemicals, including total PCBs, PAHs, and arsenic, are represented on Maps 4-14a through 4-14f as exceedances of SMS criteria. Most samples with high metals concentrations were located either downstream of RM 2.2 or generally in the same areas identified above for the other risk drivers, such as Slip 1, Slip 4, RM 1.3 to RM 1.5 on the west side, RM 2.2 on the west side, and the Boeing Plant 2/Jorgensen Forge EAA. The majority of locations with high BEHP concentrations in surface sediment were in many of these same locations (most notably in and near the Duwamish/Diagonal EAA, near RM 2.2 on the west side of

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[176] The Duwamish/Diagonal EAA was dredged and capped by King County in 2003/2004 and a thin layer of sand was placed in an area to the south and west of the EAA in 2005. In the vicinity of the Norfolk EAA, two cleanup actions have occurred. In 1999, sediments near the Norfolk CSO/SD were dredged and capped by King County. In 2003, sediment was removed from a small area adjacent to the shoreline and the area was capped by Boeing.

the LDW, Slip 4, and in the Boeing Plant 2/Jorgensen Forge EAA). Thus, to a large extent, many of the highest chemical concentrations were co-located.

10.2.2.2 Subsurface sediment patterns

Data from subsurface sediments collected from cores also provide important information and offer insight into processes that contributed to present-day distributions. Physical events, such as dredging, river channelization, and debris disposal are recorded in the geological structure of subsurface sediments. Physical and chemical markers within sediment cores can be used to reconstruct historical records of contamination and estimate the age of subsurface sediments.

Subsurface sediment chemistry data, radioisotope data, and physical characteristics of the material retained in the cores, when combined with the history of industrial development in the waterway, including chemical spills, channel diversions, and construction of the Howard Hansen Dam, allow for interpretation of the history of sedimentation in the LDW. Based on subsurface sediment stratigraphy in sediment cores, LDW sediments can be divided into the following three relatively distinct zones:

- **Recent sediment –** This upper zone represents the sediments deposited in the LDW over approximately the last 50 years and generally provides a record of the more recent activities that have occurred within the LDW since construction of the Howard Hansen Dam.
- **Upper alluvium –** The second zone, located beneath the recent sediment, represents sediments deposited in the LDW following the construction of the waterway in the early 20th century but before construction of the Howard Hansen Dam. This layer contains the record of activities that occurred within the LDW through much of the early commercial and industrial development along the LDW.
- **Lower alluvium –** The deepest zone, located beneath the upper alluvium, represents historical sediment deposits that predate construction of the LDW.

The recent sediment zone and the upper alluvium consist of material that has been deposited in the LDW since the onset of urbanization and industrial activity in the uplands surrounding the LDW.

The distribution of chemicals within cores, coupled with the physical lithology of the cores, can be used to empirically evaluate the sedimentation history for a given location. The trend of total PCB concentrations in LDW sediment cores generally reflects the introduction of PCBs (circa 1935) and the ban on the use of PCBs (1978). Peak PCB use has been recorded in Puget Sound sediment cores between 1960 and 1970 (Lefkovitz et al. 1997; Van Metre and Mahler 2004); peak total PCB concentrations are often preserved in subsurface intervals of LDW cores as well. An initial analysis was conducted to compare the depths of peak total PCB concentrations in subsurface sediment to expected depths based on the net sedimentation rates estimated using the STM and assumptions regarding the period of peak PCB usage in the LDW and minimal sediment disturbance. Most (70%) of the cores had peak concentrations as deep or deeper than expected,[177] based on the net sedimentation rates estimated by the STM and the time period of peak PCB releases. Some cores had peak total PCB concentrations that were shallower than expected; most, if not all, of these cores were either at locations with low net sedimentation rates or in areas with potential localized ongoing sources or localized disturbances, such as dredging (Maps 4-69a through 4-69d). The FS will present a more detailed analysis of MNR as a remedial option, including all lines of evidence regarding sedimentation rates.

Detected total PCB concentrations ranged from 0.52 to 890,000 µg/kg dw in subsurface sediments, with the highest concentrations generally detected below the co-located surface sediment sample or uppermost core interval, except in a few areas, including some of the EAAs, with low net sedimentation or potential ongoing sources, as discussed above (Maps 4-26a through 4-26h). Arsenic concentrations in subsurface sediment, which ranged from 3.9 to 2,000 mg/kg dw, also followed this pattern (although fewer data were available), with the highest concentrations generally detected below the co-located surface sediment sample or uppermost core interval, with a few exceptions (Maps 4-33a through 4-33e).

Profiles of cPAH concentrations in subsurface cores were less consistent. The onset of PAH contamination, typically associated with industrialization, and the decrease of PAH contamination, associated with the advent of environmental regulation from about 1972 onward, were more gradual. Moderately elevated concentrations of cPAHs were often found in uppermost core intervals, suggesting continued low-level contributions of PAHs to the LDW.

In the areas with the highest BEHP concentrations in surface sediment, such as the Duwamish/Diagonal EAA, peak BEHP concentrations were common in surface sediment or the uppermost core interval, but were otherwise variable in depth.

Dioxins and furans were generally analyzed in subsurface cores in areas with higher concentrations in surface sediment as part of the RI. In the core near RM 1.0 in the navigation channel, the highest TEQ occurred in the 4-to-6-ft interval. This TEQ was higher than the TEQ in the surface sediment sample from that area. In contrast, within the embayment between RM 1.4 and RM 1.5, which had the highest dioxin and furan TEQs in the LDW, the highest TEQ was detected in surface sediment and the uppermost core interval. In a subsurface core collected just outside the inlet, the highest TEQ in that core was detected in the 6-to-8-ft interval, but TEQs were lower. The estimated net sedimentation rate in this embayment was 2 to 3 cm/yr, although some of the embayment was not included in the model and is isolated by a pier across its mouth. The higher TEQs in surface sediment in this area indicate that burial by

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[177] When 24 cores were excluded from the analysis because they had intervals ≥ 3 ft in the portion of the core where the peak PCB concentrations were estimated or detected, 65% of the remaining cores had peak concentrations as deep or deeper than expected.

cleaner material is not occurring, potentially because of lower-than-estimated sedimentation rates in this embayment or continuing sources in this area.

Overall, 77% of the cores collected from the LDW during RI sampling events had chemical concentrations that were less than the SQS in the lowest interval of the core that was analyzed. This depth was typically reached at about 4 to 6 ft. In the cores where the lower alluvium was analyzed (50% of the cores), only one SQS exceedance was detected in that zone.

In summary, the analysis of vertical patterns of chemicals in subsurface sediment showed that peak concentrations were mostly detected in deeper core intervals in sampled areas. This observation, along with the physical sediment transport analysis, supports a general conceptual site model of net sedimentation over time at an LDW-wide scale. However, the data indicate that at a more localized scale, some areas may be affected by ongoing sources, sediment disturbances, or lower net sedimentation rates resulting in peak concentrations at the surface.

10.2.3 Exposure CSM

One of the key components of the RI is the assessment of risks for people and ecological receptors that may be exposed to site-related contaminants. This section discusses the pathways of exposure that may lead to unacceptable risks. The risk assessments are summarized in Section 10.3.

The physical and chemical CSMs discussed above provide an overview of the distribution and movement of chemicals in the LDW, which influence the exposure of receptors to these chemicals. The exposure CSM includes chemical sources, transport mechanisms, exposure pathways, exposure routes, and potentially exposed receptors. Exposure pathways were presented for ecological receptors in Figures 5-1 and 5-2 and for people in Figure 6-1. Source information and transport mechanisms were discussed in Section 9.

The importance of various exposure pathways is dependent on the chemical and the receptor of interest (e.g., people, fish), but the principal pathways for exposure to contaminated sediment in the LDW are: 1) diet, primarily through ingestion of contaminated seafood; and 2) direct contact. Other pathways of exposure were also included in the risk assessments, but were generally less important or had low risks associated with them.

10.2.3.1 Dietary Exposure

People and wildlife can be exposed to chemicals associated with sediment contamination in the LDW through ingestion of contaminated seafood, such as fish, crabs, and clams. Most resident fish, which integrate their exposure throughout their foraging ranges in the LDW, are primarily exposed through the ingestion of benthic invertebrates, which have very localized exposures, or the ingestion of other fish that have a broader exposure.

Aquatic species that serve as food for other receptors are exposed to sediment-associated chemicals through various pathways (Figures 5-1 and 5-2). Concentrations of total PCBs and arsenic in the tissues of some species (e.g., benthic invertebrates) were correlated with the concentrations of those chemicals in sediment. Weak tissue/sediment relationships were observed for arsenic and cPAHs in clams, suggesting that factors affecting arsenic and cPAH concentrations in clam tissues are complex and involve important variables that are not well understood (Sections 8.3.2 and 8.3.3).

For fish and crabs, the relationship between total PCB concentrations in sediment and tissues was modeled using a steady-state bioaccumulation model (Appendix D). This model showed that total PCB concentrations in tissues could be related to concentrations in surface sediment within acceptable uncertainty bounds. The model also suggested that as the total PCB concentrations in sediment decrease in the future, concentrations in water will exert a relatively greater effect on total PCB concentrations in tissues.

Dietary exposures in the risk assessments were estimated based primarily on consumption rates and chemical concentrations in tissues, although other factors were also included. Consumption rates for people were estimated for various scenarios. EPA and Ecology required the application of seafood consumption rates that have been developed for adult and child consumers based on seafood harvest from King County or Puget Sound. There is considerable uncertainty about the applicability of some of the seafood consumption rates to this HHRA, particularly for clams, given the quality and quantity of shellfish habitat in the LDW.

WSDOH has issued health advisories for resident fish and shellfish in the LDW, warning the public that any seafood consumption from the LDW is unsafe because of elevated total PCB concentrations in these species (WSDOH 2005). WSDOH (Office of Food Safety and Shellfish) and Public Health—Seattle and King County also advise against consumption of bivalves (i.e., clams, mussels, and oysters) from all of King County, including the LDW, because of pollution concerns (WSDOH 2005). A creel survey conducted by King County as part of their water quality assessment indicated that very few people were fishing or crabbing in the LDW (Simmonds et al. 1998). This intensive survey was conducted for 2 months in the summer of 1997, the period of highest use, and showed that use in the LDW was significantly lower than in Elliott Bay or near the Spokane Street S Bridge. However, there is a substantial Muckleshoot Tribe fishery for salmon in the LDW, and individuals have been observed trapping crabs at T-105. In addition, a relatively high number of individuals were observed fishing or crabbing from the Spokane Street S Bridge, which is located 755 ft north of the northern LDW study area boundary. The seafood consumption advisory that was in place at the time that the survey was conducted included both the LDW and the area of the Spokane Street S Bridge.[178] The home ranges of resident fish and crabs are large enough that individuals consuming fish or crabs caught at the Spokane Street S Bridge may be exposed to LDW-related contamination. Community outreach efforts conducted early in the LDW RI process indicated that the communities surrounding the LDW site were mostly unaware of this advisory (Environmental Health 2001). Since that survey was conducted, the LDW was added to the NPL (also known as the Superfund site list), and WSDOH updated their health advisories based on new data collected for the LDW RI. In addition, increased coverage by news media of the LDW has likely increased the public awareness of health risks associated with consuming resident seafood from the LDW. Future harvest rates are uncertain and are anticipated to be higher following remediation, although the availability of seafood for consumption will still be dictated largely by the availability of suitable habitat within the LDW.

To estimate exposure, chemical concentrations in tissues have been analyzed in various fish and invertebrate species that have been collected from throughout the LDW. Most of the tissue data are for English sole, shiner surfperch, Pacific staghorn sculpin, juvenile chinook salmon, Dungeness and slender crabs, soft-shell clams, and small invertebrates that live in the sediment, such as amphipods and marine worms. These species were selected because they were assumed to be representative of species within the LDW that could be consumed by people, fish, or wildlife. Their tissues were analyzed for a wide variety of chemicals.

Mean total PCB concentrations in the four sampling areas were highest for English sole, shiner surfperch, and crab hepatopancreas tissue (concentrations in edible crab meat were lower). Mean total PCB concentrations were lowest in mussels. Mean inorganic arsenic concentrations were highest in soft-shell clams. The highest cPAH concentrations in tissue were detected in clams, mussels, and benthic invertebrates. With the exception of one staghorn sculpin sample, concentrations were not as high in fish because fish metabolize PAHs. The concentration in one staghorn sculpin sample was similar to concentrations in some of the invertebrate samples. Other SVOCs were infrequently detected in tissue. DDT and other chlorinated pesticides were detected in tissue samples, but there is high uncertainty in these data because of analytical interference from PCBs. The highest mean concentrations of metals (including TBT) were detected in invertebrates.

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[178] Prior to the 2005 seafood consumption advisory update published by Washington Department of Health (WSDOH 2005), Public Health – Seattle & King County had an existing seafood consumption advisory for urban areas along the King County shoreline, including Elliott Bay and the LDW. The advisory warned of contaminants in shellfish, crabs, and bottom fish, but did not provide consumption limits, nor did it give specific advice against eating any of these species.

10.2.3.2 Direct contact

People are also exposed to chemicals in contaminated sediment through direct contact. The HHRA evaluated several scenarios for this pathway, including beach play, clamming, and netfishing, where people would come into direct contact with sediment. While many ecological species can be exposed through direct contact, the benthic invertebrate community is most closely related to direct sediment exposure pathways because they are constantly in direct contact with sediment.

10.3 RISK ASSESSMENT RESULTS

Baseline risk assessments were conducted to evaluate the potential for adverse effects for people and ecological species that may be exposed directly or indirectly to contaminated sediment in the LDW, as discussed above. Key findings from these assessments are presented in this section. The risk assessments are presented in full in Appendices A and B and summarized in Sections 5 and 6.

10.3.1 Ecological Health

The ERA assessed risks to a number of receptor groups that were selected to be both representative and protective of the range of species found in the LDW, either as resident species or transient/migratory species. Receptor groups included the benthic invertebrate community (including gastropods), fish, crabs, birds, and mammals. A summary of COCs based on the ERA is presented in Table 10-1.

Table 10-1. Summary of COCs based on the baseline ERA

COC	BENTHIC INVERTEBRATE COMMUNITY[a]	FISH	SPOTTED SANDPIPER	GREAT BLUE HERON	OSPREY	RIVER OTTER	HARBOR SEAL
PCBs	X	X	X			X	
Cadmium	X	X					
Chromium	X		X				
Copper	X		X				
Lead	X		X				
Mercury	X		X				
Vanadium		X	X				
BEHP	X						
Zinc	X						
Silver	X						
BBP	X						
Fluoranthene	X						
Indeno(1,2,3-cd)pyrene	X						
Arsenic	X						

[a] Forty-one chemicals were considered to be COCs and risk drivers for the benthic invertebrate community, based on exceedances of the chemical criteria of the SMS in one or more surface sediment samples within the LDW. Three additional COCs were identified with concentrations higher than TRVs or toxicologically based DMMP guidelines. The full list is presented in Table 5-6. COCs with eight or more CSL exceedances in surface sediments are included in this table.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
DMMP – Dredged Material Management Program
COC – chemical of concern
ERA – ecological risk assessment
PCB – polychlorinated biphenyl
TRV – toxicity reference value

Underlined Xs signify risk drivers.

Overall, seven chemicals were identified as COCs (total PCBs, cadmium, chromium, copper, lead, mercury, and vanadium) for at least one fish or wildlife species (Table 10-1). COCs were defined as chemicals with exposure levels that exceeded the lowest adverse effects thresholds. For most ecological species, risks were estimated to be low. Somewhat higher risks were estimated for river otter based on PCB exposures and for spotted sandpiper based on exposure to lead in one EAA (Table 5-16). No quantitative ecological risk estimates were calculated for dioxins and furans and fish or wildlife species because tissue data were not available when the ERA was completed.

For the benthic invertebrate community, 41 COCs were identified based on at least one surface sediment sample exceeding its SQS; three additional COCs were also identified with concentrations higher than TRVs or toxicologically based DMMP guidelines. Single locations with COC concentrations above the SQS for one or more chemicals were found throughout the LDW. Sediment toxicity tests were conducted primarily at locations where chemical concentrations suggested adverse impacts to benthic invertebrates could occur. Of the 48 sediment samples tested for toxicity during the RI, 11 exceeded the SQS biological effects criteria, 19 exceeded the CSL biological effects criteria, and 18 did not exceed either the SQS or CSL biological effects criteria. Based on sediment chemistry and toxicity data, no adverse effects on the benthic invertebrate community are expected for approximately 75% (345 ac) of the overall LDW area (i.e., the area in which chemical concentrations were less than or equal to chemical SQS criteria and where sediments were non-toxic[179] according to SQS biological criteria; Map 4-16).

There is a higher likelihood for adverse effects in approximately 7% of the LDW area (34 ac), which was found to have chemical concentrations or biological effects in excess of the CSL. The remaining 18% of the LDW area (82 ac) had chemical concentrations or biological effects between the SQS and CSL, indicating that risks to the benthic invertebrate community are less certain than in areas with concentrations greater than

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[179] Sediment samples from 46 locations with chemical SQS exceedances were tested for toxicity. Sixteen of the samples were not toxic relative to SQS biological effects criteria. Toxicity tests were generally not conducted in areas with the highest contamination (e.g., EAAs), which were assumed to be toxic.

one or more CSL. Some areas had multiple locations with chemical concentrations greater than the CSL. Most of these areas are located within or near the EAAs.

Total PCBs and BEHP were the chemicals that most frequently exceeded the SQS and the CSL in surface sediment samples. Over 60% of the SQS exceedances for both total PCBs and BEHP and 71 and 90% of BEHP and total PCB CSL exceedances, respectively, were in EAAs. Mercury was the only other chemical with more than 20 locations with CSL exceedances. Other chemicals with more than 20 locations with SQS exceedances were butyl benzyl phthalate, lead, mercury, zinc, and a few individual PAHs. Risks to the benthic invertebrate community from TBT were low.

10.3.2 Human health

For the HHRA, risks were evaluated for several different seafood consumption scenarios and also for direct sediment exposure scenarios associated with beach play, clamming, or netfishing. There are limited data available on the amount of resident seafood organisms currently being harvested and consumed from the LDW. In the absence of site-specific human seafood consumption rates, EPA and Ecology required the application of seafood consumption rates that have been developed for tribal consumers based on seafood harvest from other areas of Puget Sound. The tribal seafood consumption rates represent relatively high rates that might occur for harvest from the LDW at some time in the future, although they likely overestimate the current consumption of resident seafood organisms from the LDW. In addition, API consumption rates were based on rates derived from self-harvested seafood from throughout King County (not limited to the LDW). It is not known if the seafood resources within the LDW at some future time would be sufficient to support these consumption rates, but these rates were assumed to represent hypothetical consumption rates for the estimation of risks.

Similarly, there are no LDW-specific data to estimate the degree to which humans may currently be directly exposed to sediments via beach play or clamming activities. The exposure scenarios assumed for these activities were intended to represent exposures for a health-protective estimation of risks. The tribal netfishing scenario, on the other hand, reflects exposure conditions that could occur under current tribal fishing activities within the LDW.

Using health-protective exposure assumptions, most of the risks are associated with a few chemicals (i.e., PCBs, arsenic, cPAHs, and dioxins and furans), which were designated as risk drivers (Table 10-2). Estimated excess cancer risks in the LDW were found to be highest for the seafood consumption scenarios. There were distinct differences in the contributions from each seafood type to the risk estimate for each chemical. For total PCBs, the excess cancer risks for the tribal adult RME scenario were attributed to several seafood types, including clams (35%), crabs (29%), and pelagic fish (24%). For arsenic and cPAHs, approximately 95% of the risk was attributed to consumption of clams.

Table 10-2. Summary of COCs based on the baseline HHRA

COC[a]	SEAFOOD CONSUMPTION	CLAMMING	BEACH PLAY	NETFISHING
Arsenic (inorganic)	X	X	X	X
cPAHs	X	X	X	X
Total PCBs	X	X	X	X
Dioxins and furans	X[b]	X	X	X
Aldrin	X			
BEHP	X			
alpha-BHC	X			
beta-BHC	X			
Carbazole	X			
Total chlordane	X			
Total DDTs	X			
Dieldrin	X			
gamma-BHC	X			
Heptachlor	X			
Heptachlor epoxide	X			
Hexachlorobenzene	X			
Pentachlorophenol	X			
Toxaphene		X		X
TBT	X			
Vanadium	X			

[a] COCs were defined in the HHRA based on risk estimates only from RME scenarios.

[b] Dioxins and furans were assumed to be a risk driver in the absence of any tissue chemistry data based on an expectation that if such data had been collected, the risks would have been unacceptable.

BEHP – bis(2-ethylhexyl) phthalate
BHC – benzene hexachloride
COC – chemical of concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
HHRA – human health risk assessment
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
TBT – tributyltin

Underlined Xs signify risk drivers.

Excess cancer risks for direct sediment contact RME scenarios for netfishing, clamming, and beach play were much lower than risk estimates for the seafood consumption scenarios. Most risks for beach play and clamming were associated with arsenic and cPAHs; for netfishing, the highest risks were associated with dioxins and furans. None of the direct sediment contact RME scenarios were estimated to result in non-cancer health risks.

10.4 RISK MANAGEMENT CONSIDERATIONS

This section summarizes key findings of the RI that are related to ongoing risk management decisions that will be made for the site. Key findings are presented for risk-based threshold concentrations and preliminary background concentrations of the risk drivers, and the relationship between the spatial distribution of risk driver chemicals in sediment and LDW sediment dynamics.

10.4.1 RBTCs and chemical concentrations in areas outside of the LDW

Risk estimates presented in the baseline risk assessments have been translated into sediment RBTCs that will be considered in the FS in combination with background concentrations and other factors. Sediment RBTCs were derived for direct sediment contact scenarios and also for seafood ingestion scenarios for humans and river otters, as appropriate, for four risk driver chemicals (total PCBs, arsenic, cPAHs, and dioxins and furans) (Table 10-3). Sediment RBTCs for benthic invertebrates were set equal to SMS criteria.

Information was compiled on concentrations of four risk driver chemicals (total PCBs, arsenic, cPAHs, and dioxins and furans) in areas outside of the LDW (Section 7). Background concentrations provide important context for RBTCs because EPA generally does not require cleanup below anthropogenic background concentrations in consideration of the potential for recontamination from sources unrelated to the site, cost effectiveness, and technical practicability (EPA 2002d). Under MTCA, natural background concentrations[180] are used in the evaluation of sediment cleanup levels as a lower limit below which cleanup levels cannot be achieved (WAC 173-340-705(6)).

Background concentrations will be considered in the FS and their role in determining cleanup levels will be determined by EPA and Ecology in the ROD. Concentrations of risk driver chemicals in Puget Sound reference areas are presented in Section 7.1 and summarized in Table 10-3. Relevant data for upstream environments are presented in Section 7.2 and Table 10-3. In addition to the upstream inputs, other anthropogenic inputs from the surrounding urban area could enter the LDW through atmospheric deposition and stormwater runoff, but these inputs were not quantified as part of the RI studies.

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[180] According to MTCA (WAC 173-340-200), natural background concentrations are concentrations of hazardous substances that are consistently present in an environment that has not been influenced by localized human activities.

Table 10-3. Comparison of sediment RBTCs and concentrations for selected risk driver chemicals in samples collected from Puget Sound reference areas, upstream of the LDW and the Upper Turning Basin

Risk Driver Chemical	Unit	Risk Level	Sediment RBTCs for RME Direct-Contact Scenarios (Tribal Clamming, Beach Play, Netfishing)	Sediment RBTCs for RME Seafood Consumption Scenarios	Surface Sediment Data		Subsurface Sediment Data	Water Data
					Puget Sound Reference Areas[a]	Upstream[b]	Upper Turning Basin[c]	Upstream Suspended Solids[d]
Arsenic	mg/kg dw	1×10^{-6}	1.3, 2.8, 3.7[e]	nc[f]	2.3 – 15.9	11	13	73
		1×10^{-5}	13, 28, 37[e]	nc[f]				
		1×10^{-4}	130, 280, 370[e]	nc[f]				
		HQ = 1	na	na				
		SQS	57	na				
		CSL	93	na				
cPAHs[g]	µg/kg dw	1×10^{-6}	150, 90, 380[e]	nc[h]	14.7 – 244.7	135	201	354
		1×10^{-5}	1,500, 900, 3,800[e]	nc[h]				
		1×10^{-4}	15,000, 9,000, 38,000[e]	nc[h]				
		HQ = 1	na	na				
Dioxin and furan TEQ	ng/kg dw	1×10^{-6}	13, 28, 37[e]	nc[i]	0.141 – 2.3	2.0[j]	no data	no data
		1×10^{-5}	130, 280, 370[e]	nc[j]				
		1×10^{-4}	1,300, 2,800, 3,700[e]	nc[j]				
		HQ = 1	na	na				

Table 10-3, cont. Comparison of sediment RBTCs and concentrations for selected risk driver chemicals in samples collected from Puget Sound reference areas, upstream of the LDW and the Upper Turning Basin

Risk Driver Chemical	Unit	Risk Level	Sediment RBTCs for RME Direct-Contact Scenarios (Tribal Clamming, Beach Play, Netfishing)	Sediment RBTCs for RME Seafood Consumption Scenarios	Surface Sediment Data		Subsurface Sediment Data	Water Data
					Puget Sound Reference Areas[a]	Upstream[b]	Upper Turning Basin[c]	Upstream Suspended Solids[d]
Total PCBs	µg/kg dw	1×10^{-6}	500, 1,700, 1,300[e]	< 1[k]	0.2 – 19.9	40	86	107
		1×10^{-5}	5,000, 17,000, 13,000[e]	< 1[k]				
		1×10^{-4}	50,000, 170,000, 130,000[e]	7.3 – 185[l]				
		HQ = 1	nc[m]	128 – 159[n]				
		SQS	130[o]	na				
		CSL	1,000[p]	na				

[a] Information relevant to calculating preliminary natural background concentrations in sediment is presented in Section 7.1. The concentration ranges presented in this table (for informational purposes only) represent the ranges of 90th percentile concentrations calculated for multiple Puget Sound reference areas.

[b] Information relevant to upstream concentrations in sediment is presented in Section 7.2. The concentrations presented in this table (for informational purposes only) represent the 90th percentile concentrations in the upstream and Upper Turning Basin datasets. Other anthropogenic inputs from the surrounding urban area are likely due to atmospheric deposition and stormwater runoff but were not quantified as part of the RI studies. Concentrations in urban bays and lakes are discussed in Section 7.3.

[c] Information relevant to concentrations in subsurface sediment from the Upper Turning Basin is presented in Section 7.2.3. The concentrations presented in this table (for informational purposes only) represent the 90th percentile concentrations in sediment cores collected between RM 4.0 and RM 4.75.

[d] Information relevant to concentrations in suspended solids is presented in Section 7.2.4. For PCBs and cPAHs, suspended solids concentrations were estimated by normalizing whole-water concentrations in upstream samples to the sample-specific total TSS concentrations. For arsenic, sample-specific total and dissolved concentrations and TSS in upstream water samples were used to estimate suspended solids concentrations. The concentrations presented in this table (for informational purposes only) represent the 90th percentile concentrations.

[e] Sediment RBTCs developed from the human health direct contact RME scenarios (i.e., tribal clamming, beach play, netfishing).

Although identified as a COC for seafood consumption, sediment RBTCs were not calculated for arsenic for seafood consumption scenarios because a clear relationship has not been established between arsenic concentrations in surface sediment and clam tissues, but would be needed to develop an RBTC.

[g] cPAH concentrations are expressed in terms of benzo(a)pyrene equivalents.

[h] Although identified as a COC for seafood consumption, sediment RBTCs were not calculated for cPAHs for seafood consumption scenarios because a clear relationship has not been established between cPAH concentrations in surface sediment and clam tissues, but would be needed to develop an RBTC.

[i] Although identified as a COC for seafood consumption, sediment RBTCs were not calculated for dioxins and furans for seafood consumption scenarios because no tissue data were available from the LDW at the time of the risk assessments.

[j] Because of the small dataset, 90th percentiles were not calculated for dioxin and furan TEQs. The dioxin and furan TEQ represents the mean TEQ in upstream surface sediment samples. [f]

[k] A sediment RBTC could not be calculated; even if the total PCB concentration in sediment was set equal to 0 µg/kg dw, FWM-estimated total PCB concentrations in tissue would be greater than the tissue RBTC for the applicable risk level because of the contribution of PCBs from water alone.

Table 10-3, cont. Comparison of sediment RBTCs and concentrations for selected risk driver chemicals in samples collected from Puget Sound reference areas, upstream of the LDW and the Upper Turning Basin

[l] Sediment RBTCs developed from the human health seafood consumption RME scenarios.

[m] Sediment RBTCs were not estimated for non-cancer hazards for direct-contact scenarios because none of the RME scenarios had HQs for an individual chemical greater than 1 or generated endpoint-specific HIs in excess of 1 (Appendix B, Section B.5.6).

[n] RBTCs developed from the otter prey ingestion scenario, estimated through the FWM.

[o] Reported value is lowest AET, which is functionally equivalent to the SQS, but in dry weight units.

[p] Reported value is second lowest AET, which is functionally equivalent to the CSL but in dry weight units.

AET – apparent effects threshold
COC- chemical of concern
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CSL – cleanup screening level
dw – dry weight
FWM – food web model
HI – non-cancer hazard index
HQ – non-cancer hazard quotient
LDW – Lower Duwamish Waterway
na – not available or applicable
nc – not calculated
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RI – remedial investigation
RM – river mile
RME – reasonable maximum exposure
SQS – sediment quality standards
TEQ – toxicity equivalent quotient
TSS – total suspended solids

Sediment RBTCs at the 1 × 10^{-6} level for direct sediment contact are higher than the range of Puget Sound reference area concentrations for total PCBs and dioxins and furans (Table 10-3). The cPAH sediment RBTC for netfishing at the 1 × 10^{-6} risk level is also greater than the range of Puget Sound reference area concentrations. The sediment RBTCs for cPAHs at the 1 × 10^{-6} threshold for beach play and clamming scenarios are within the range of Puget Sound reference area concentrations. For arsenic, the direct sediment contact RBTCs at the 1 × 10^{-6} risk level are toward the lower end or below the range of the Puget Sound reference area concentrations.[181]

RBTCs derived for the direct sediment contact scenarios should be applied to sediment concentrations averaged over the exposure areas for the specified uses, not on a point-by-point basis. The exposure area for the netfishing RME scenario included all of the intertidal and subtidal areas of the LDW. For the beach play RME scenario, eight separate intertidal areas were selected in the HHRA as exposure areas where children would be most likely to contact sediment (Map 6-1). These areas could change in the future if, for example, a new park was constructed. The exposure area for the RME clamming scenario included all intertidal areas where clamming could occur because such areas are potentially accessible either from a boat or from shore (Map 6-2).

Sediment RBTCs for total PCBs were estimated using a FWM calibrated to LDW environmental conditions and species for the RME human seafood consumption scenarios and for river otters, which had risk estimates greater than 1 × 10^{-6} or LOAEL-based HQs > 1, respectively. Sediment RBTCs estimated for human seafood consumption scenarios and for river otter prey ingestion are also intended to be average concentrations over the relevant exposure area, and not applied on a point-by-point basis. Sediment RBTCs for cPAHs and arsenic associated with the human consumption of clams could not be derived because a clear relationship has not been established between concentrations of these chemicals in clams and in sediment (Sections 8.3.2 and 8.3.3). Sediment RBTCs for dioxins and furans were not derived for seafood consumption pathways because tissue data were not available from the LDW at the time of the risk assessments (Section 8.3).

A wide range of sediment RBTCs for total PCBs was derived for the seafood consumption scenarios (Tables 8-10 and 8-11). Very few sediment RBTCs (and none of the sediment RBTCs for the RME scenarios) could be derived for the 1 × 10^{-6} risk level because the contribution from water alone resulted in estimated total PCB concentrations in tissue greater than this risk level, even in the absence of any

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[181] Of the 10 Puget Sound reference areas, one area (Rich Passage) had a 90th percentile arsenic concentration of 2.3 mg/kg dw; the remaining nine areas had 90th percentile arsenic concentrations ranging from 6.3 to 15.9 mg/kg dw. At higher risk levels, direct-contact sediment RBTCs are above the concentration ranges of samples collected from Puget Sound reference areas, upstream, and the Upper Turning Basin, except for arsenic. The direct-contact sediment RBTC for tribal clamming for arsenic at the 1 × 10^{-5} risk level is within the range of these concentrations.

contribution from sediment. The sediment RBTCs for non-RME scenarios that could be derived for the 1 × 10^{-6} risk level were lower than the range of the Puget Sound reference area concentrations for total PCBs for all non-RME scenarios, except the sediment RBTCs for the API CT scenario (10 µg/kg dw) and for the one-meal-per-month consumption of clams scenario (2 µg/kg dw), which were in the range of Puget Sound reference area concentrations (0.2 to 19.9 µg/kg dw).

At the 1 × 10^{-5} risk level, sediment RBTCs were still < 1 µg/kg dw for all of the RME seafood consumption scenarios, the adult tribal (Suquamish data) scenario, and the one-meal-per-month consumption of pelagic fish scenario. However, 1 × 10^{-5} sediment RBTCs calculated for other non-RME scenarios were generally higher than the range of the Puget Sound reference area concentrations, except for the RBTC for the one-meal-per-month consumption of benthic fish (13 µg/kg dw), which was within the 0.2 to 19.9 µg/kg dw range found in Puget Sound reference areas.

At the 1 × 10^{-4} risk level, sediment RBTCs for total PCBs could be calculated for all of the seafood consumption scenarios, except the non-RME adult tribal (Suquamish data) scenario. All of the calculated sediment RBTCs were greater than the Puget Sound reference area concentrations and greater than or within the range of the upstream and Upper Turning Basin (subsurface sediment) concentrations, except for the sediment RBTCs for the adult tribal RME scenario (Tulalip data; 7.3 µg/kg dw) and the adult tribal non-RME scenario (Suquamish data; < 1 µg/kg dw) (Tables 8-10 and 8-11).

Tissue RBTCs for total PCBs were also calculated for a given risk level and seafood consumption rate. For example, the total PCB concentration in seafood that would be necessary to achieve an excess cancer risk of 1 × 10^{-5} would be 4.2 µg/kg ww for the adult tribal RME scenario (Table 10-4) and 14 µg/kg ww for the API RME scenario. Just as it is important to put sediment RBTCs in the context of sediment background levels, it is important to put tissue RBTCs in the context of the PCB concentrations in other common dietary food items. As described in Section 7.4.1 and in WSDOH (2006), PCBs are frequently detected in many different seafood species throughout non-urban areas of Puget Sound at concentrations similar to tissue RBTCs, with mean total PCB concentrations ranging from 4.5 to 32.3 µg/kg ww in rockfish, 2.8 to 11.6 µg/kg ww in English sole fillet, 0.62 to 25 µg/kg ww in crab edible meat and hepatopancreas, and 0.12 to 2.8 µg/kg ww in clams (Table 10-4). Consumption of English sole, crabs, or clams from non-urban areas of Puget Sound at a rate equivalent to the adult tribal RME scenario would yield excess cancer risk estimates of 1 × 10^{-6} or higher, based on the total PCB concentrations in these tissues (Figures 10-5, 10-6, and 10-7). In addition, these tissue RBTCs are lower than the PCB concentrations that have been detected in some common food items from a typical diet (WSDOH 2006). Consumption of many of these common food items at the rates assumed under these RME scenarios would subject the consumer to risks that would be deemed unacceptable in the MTCA and CERLCA context. Tissue RBTCs for lower consumption rates are much higher than tissue RBTCs for the RME scenarios, with the exception of the 1 × 10^{-6} tissue RBTC for the one-meal-per-month scenario (11 µg/kg ww), because of the large differences among the consumption rates.

Table 10-4. Total PCB concentrations in LDW and greater Puget Sound tissue compared with tissue RBTCs

EPC OR RBTC	TOTAL PCB CONCENTRATION (µg/kg ww)		
	ENGLISH SOLE	CLAM	DUNGENESS AND SLENDER CRAB
LDW-wide UCL	1,200	600	200
LDW-wide mean	700	140	170
Greater Puget Sound (range of mean values)[a]	2.8 – 11.6[b]	0.12 –- 2.8[c]	0.62 – 8.44[d]
1 × 10^{-4} RBTC (Tulalip RME)	42	42	42
1 × 10^{-5} RBTC (Tulalip RME)	4.2	4.2	4.2
1 × 10^{-6} RBTC (Tulalip RME)	0.42	0.42	0.42

[a] Range of mean total PCB concentrations from various studies and non-urban sampling areas in the greater Puget Sound, as presented in Table 7-15.

[b] Total PCB concentrations in English sole fillet tissue.

[c] Total PCB concentrations in clam soft tissue.

[d] Total PCB concentrations in crab edible meat or calculated whole-body tissue.

EPC – exposure point concentration
LDW – Lower Duwamish Waterway
PCB – polychlorinated biphenyl
RME – reasonable maximum exposure
RBTC – risk-based threshold concentration
UCL – upper confidence limit on the mean
ww – wet weight

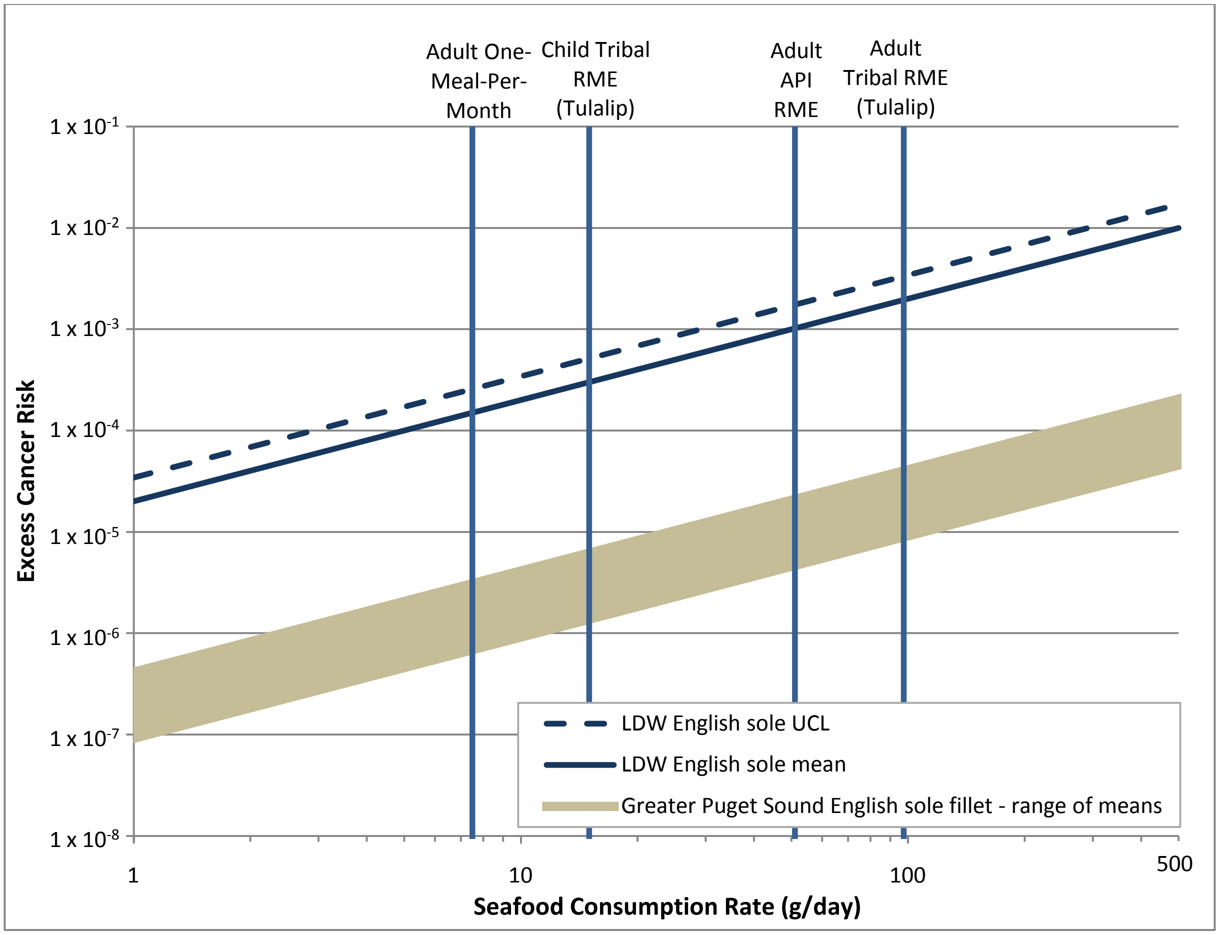
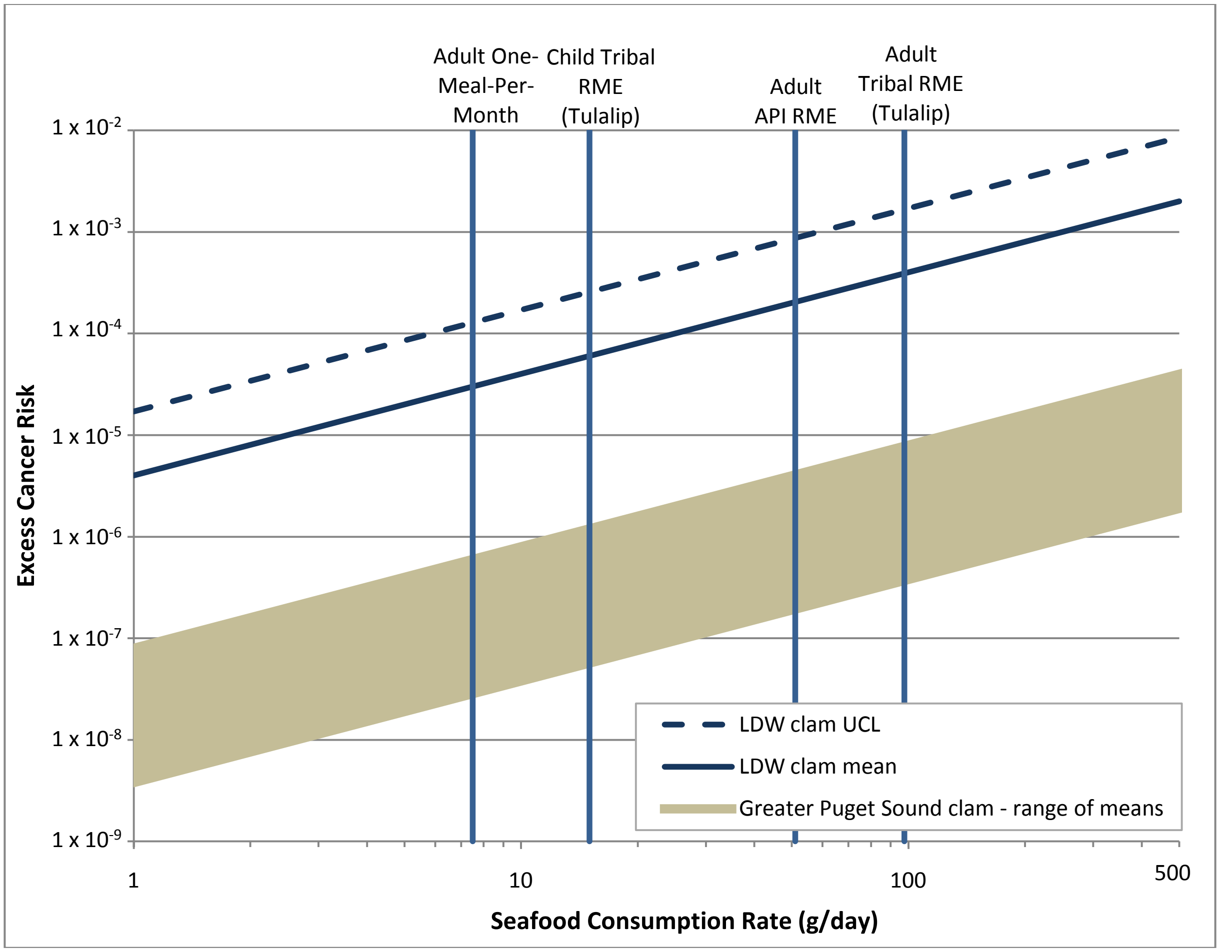


Figure 10-5. **Excess cancer risks calculated using total PCB concentrations in English sole fillet composite samples collected from the LDW and from greater Puget Sound locations as a function of seafood consumption rate**



Note: Clam samples from greater Puget Sound locations included geoduck from Freshwater Bay, horse clam tissue (excluding the visceral cavity) from Freshwater and Dungeness Bays, and littleneck clams from Padilla and Fidalgo Bays. None of these species are directly comparable to *Mya arenaria*, the soft-shelled clams sampled in the LDW.

Figure 10-6. Excess cancer risks calculated using total PCB concentrations in clam tissue composite samples collected from the LDW and from greater Puget Sound locations as a function of seafood consumption rate

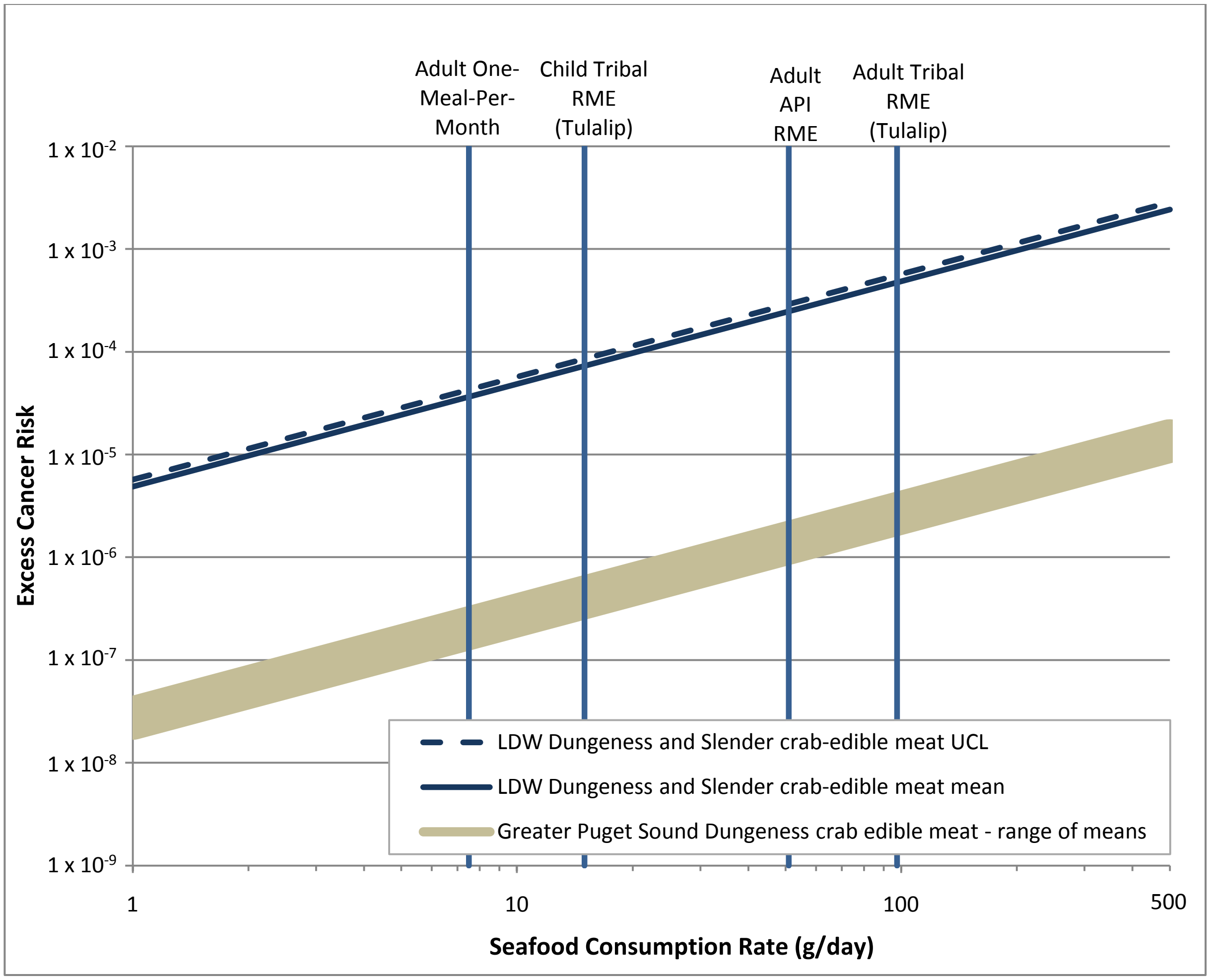


Figure 10-7. Excess cancer risks calculated using total PCB concentrations in crab edible meat composite samples collected from the LDW and from greater Puget Sound locations as a function of seafood consumption rate

10.4.2 Implications of chemical patterns and trends

As presented in Section 4 and further discussed in Section 10.2.2, chemicals in LDW sediments exhibit distinct patterns and trends. Chemical patterns that are of interest include spatial distributions in surface sediments, where exposure to ecological receptors and humans is most likely, and vertical distributions, which provides a record of the past and an indication of the rate of net sedimentation that has resulted in the burial of historical surface sediments. The presence of peak total PCB concentrations below the sediment surface at many different locations throughout the entire site is consistent with the independent lines of evidence that indicate that the LDW is largely a net depositional environment and that contamination associated with historical releases is being buried in most locations, with a few notable exceptions. These results will be further evaluated in the FS.

Despite the prevalence of peak total PCB concentrations at depth, elevated concentrations were also detected in surface sediments in distinct areas. The locations of these elevated concentrations were predominantly in areas where elevated concentrations were also detected at depth and historical contamination remains at the surface because of low net sedimentation rates in these areas. In other areas, the presence of elevated concentrations in surface sediments may be evidence of ongoing localized sources or localized disturbances. Focused sampling and source control evaluations at some of the EAAs confirm that ongoing sources are the most likely explanation for the presence of elevated concentrations at some of the EAAs.

The STM and the physical CSM are useful for evaluating future exposure conditions, especially in areas outside the EAAs. Sediment particles will be continually transported into the LDW, predominantly from the Green River, which is the source of approximately 99% of the sediments depositing within the LDW. Consequently, surface sediment quality in the LDW will eventually resemble the quality of the particles originating from the Green River, assuming that local sources are sufficiently controlled and areas with elevated concentrations and low sediment deposition rates are remediated. Results from the STM model indicate that 90% of the LDW has an estimated net sedimentation rate of 1.0 cm/yr or greater (Map 3-9). Thus, within 10 years, all of this area would receive at least 10 cm of new sediment, almost all of which comes from material flowing into the LDW from the Green River. The significance of these results, and the scale at which the predictive model is valid for estimating recovery, will be evaluated in the FS.

The Slip 4 area is instructive when considering the roles both net deposition and local sources can have on the distribution of risk drivers. This area has been intensively sampled recently in preparation for a non-time critical removal action. Slip 4 is within an area that has estimated net sedimentation rates between 1 and 3 cm/yr. Map 4-23 presents a time series of total PCB concentrations in surface sediment within Slip 4. There has been a marked decrease in surface sediment concentrations between 1990 and 2004 for much of the slip. This decrease suggests that chemical concentrations in depositing sediment (which includes sediment from upstream as well as from point sources such as storm drains) have decreased sufficiently to improve surface sediment quality during this period. The area of Slip 4 where elevated concentrations of total PCBs persist, even though at relatively lower concentrations, is in the head of the slip where there are active storm drain outfalls. Source identification studies in the storm drain systems have indicated ongoing sources of PCBs, which (until addressed) have the potential for limiting further improvements in sediment quality or recontaminating the slip after the removal action. This example illustrates the integrated temporal and spatial patterns that can be expected in depositional areas of the larger LDW.

The overall implication is that both active source control and remedial actions are likely to be required to reduce overall risks in the LDW. Focused investigations in the EAAs have highlighted the need to implement control measures for ongoing sources prior to remediation, while time trend data from Slip 4, for example, suggest that continuing inputs of sediment from the Green River will result in improvements in surface sediment quality over time, except where additional efforts at source control are required.

10.5 ONGOING SOURCE CONTROL EFFORTS

Controlling sources is critical to the long-term success of any remedial action (EPA 2002c). Consistent with this concept, Ecology (2007j) has developed a source control strategy for the LDW which is currently being implemented. This program is being conducted in parallel with the RI/FS and is expected to continue into the foreseeable future as remedial actions are undertaken, where needed, to reduce human health and environmental risks. Source identification and source control actions will need to precede remediation or these areas may become recontaminated.

Initial source control efforts have been focused on source identification at those EAAs that are being remediated as non-time critical removal actions. Ecology, along with the SCWG, is leading the source control evaluation. As part of their source control evaluation process, data gaps reports and SCAPs are being prepared by Ecology to help direct future investigation and remediation efforts while assessing potential sources for the chemicals identified. These efforts will continue to evaluate potential sources to the LDW and implement corrective actions to reduce or eliminate identified sources.

In addition, current regulatory requirements have established inspection and compliance programs to monitor and control handling and disposal of manufacturing wastes produced by industrial operations along the LDW. These regulations have helped reduce sources of contamination that, in past practices, might have resulted in releases to the LDW. Permit programs are in place to help monitor and manage direct discharges to the LDW from CSOs, EOFs, and SDs in the basin. For example, since 1970, the City of Seattle has reduced the annual CSO volume citywide by approximately 95%, and since 1988, King County has reduced their annual CSO volume by 63%.

Continued coordination of the CERCLA/MTCA cleanup actions and source control programs (both the SCWG and other regulatory programs) will ensure that future remedial actions will not be unduly impacted by local sources.

10.6 KEY OBSERVATIONS AND FINDINGS

Key observations and findings for the RI are summarized below.

- Over the past 100 years, the LDW has been highly modified from its natural configuration to support urban and industrial development. Changes have included reductions and control of water flow, significant shoreline modifications, loss of intertidal habitat, and installation of riprap, pier aprons and sheet pile walls. Some limited areas of natural shoreline still exist within the LDW.

- Industrial and commercial facilities occupy most of the shoreline; one residential community is also located along the shoreline with another community nearby.
- The LDW is currently used as an industrial navigational corridor. It also supports recreational uses such as boating, kayaking, fishing, and beach play. The LDW is also one of the locations of the Muckleshoot Tribe's commercial, ceremonial, and subsistence fishery for salmon, and the Suquamish Tribe actively manages aquatic resources north of the Spokane Street Bridge, located just north of the LDW. The Duwamish Tribe uses Herring's House Park and other parks along the Duwamish for cultural gatherings.
- Despite significant alterations in habitat and areas with elevated chemical concentrations, the LDW contains a diverse assemblage of aquatic and wildlife species and a robust food web that includes top predators.
- The majority of the high arsenic and total PCB concentrations in surface sediment were located within fairly well-defined areas. The locations of the highest arsenic and total PCB concentrations were generally not in the same areas, indicating that sources likely differ for these two chemicals. Areas with the highest cPAH concentrations were located in many of the same areas identified for arsenic and total PCBs, but were also more dispersed. There are several areas with high dioxin and furan TEQs in surface sediments.
- Most of the human health risk is from PCBs, arsenic, cPAHs, and dioxins and furans.
- The highest risks to people are associated with consumption of fish, crabs, and clams, with lower risks associated with activities that involve direct contact with sediment, such as clamming, beach play, and netfishing.
- Ecological risks to fish and wildlife were relatively low, with the exception of risks to river otter from PCBs.
- Based on sediment chemistry and toxicity test results, sediment contamination in approximately 75% of the LDW (345 ac) is estimated to have no effect on the benthic invertebrate community; approximately 7% (34 ac) of the surface sediment has chemical concentrations exceeding the higher of the two state standards, and is therefore expected to have adverse effects on the benthic invertebrate community (see WAC 173-204-310[b]). The remaining 18% of the LDW (82 ac) has chemical concentrations exceeding the lower of the two state standards and these areas are considered to have minor adverse effects on the benthic invertebrate community. Most of the exceedances of SMS criteria were for PCBs and phthalates, although 41 different chemicals had at least one exceedance.

- Sediment is continually depositing within the LDW, with almost all new sediment (99%) originating from the Green River. The STM estimates that over 200,000 metric tons of sediment per year enter the LDW. Approximately 50% of this load deposits in the LDW. STM modeling runs indicate that approximately 90% of the total bed area in the LDW receives 10 cm of new sediment (from the combined Green River and lateral sources) within 10 years or less. This sediment is mixed with the existing bedded sediment through various processes, including bioturbation and propeller wash.
- A few areas in the LDW will be scoured during high-flow events. Based on the STM, the maximum scour depth is relatively shallow and is generally limited to sediment in the top 20 cm; thus, deeper sediments would not be exposed as a result of high-flow events. Scour to these relatively shallow depths is expected to occur in relatively small areas of the LDW. The STM did not account for scour from localized activities, such as discharges from outfall, tugboat maneuvering, or anchor dragging, that could have caused localized erosional environments. Routine boat traffic is expected to mix the top few cm of sediment, which is part of the biologically active zone mixed by benthic invertebrates, whereas tugboat maneuvering is a potential source of localized erosion that could disturb sediment at greater depths in small areas. In addition, in some areas, ships may have caused localized erosion from physical forces (i.e., anchor dragging) unrelated to propeller-driven scour. Site-specific information, in addition to the results of the STM, will be evaluated in any future remedial designs.
- The physical conceptual site model of a net depositional environment is supported by both physical and chemical lines of evidence, including lithology and chemistry profiles in sediment, where these data were collected. Empirically derived net sedimentation rates from chemical and physical markers were within 1 cm/yr of the modeled sedimentation rates for 40 of 63 core locations. Most (70%)[182] of the 157 core locations evaluated had depths of peak PCB concentrations that were consistent with those predicted by the STM, with peak PCB concentrations as deep or deeper than those estimated by the STM. The remaining 30% of the cores had peak concentrations that were shallower than expected, assuming net sedimentation rates from the STM, peak PCB usage and release in the 1960s and 1970s, and minimal localized disturbances.
- Based on the STM, LDW surface sediment is generally expected to become more similar in character over time to the sediment being transported by the

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[182] The percent of cores with peak concentrations as deep or deeper than expected is 65% when 24 cores were excluded from the analysis because they had samples composited over intervals ≥ 3 ft in the portion of the core where the peak PCB concentrations were estimated or detected.

Green River on an LDW-wide scale; localized areas may continue to be influenced by inputs from sources in those areas.

- A comparison of background concentrations with risk-based goals in sediment (represented by sediment RBTCs) will be important in risk management decisions by EPA and Ecology. Puget Sound sediment data will be considered in the derivation of natural background concentrations and data from other lines of evidence will be considered in the derivation of anthropogenic background concentrations.
- For total PCBs, all sediment RBTCs for direct-contact scenarios are above the concentration range of samples collected from Puget Sound reference areas, upstream of the LDW, and the Upper Turning Basin (subsurface sediment). All sediment RBTCs for RME human seafood consumption scenarios are below the range of Puget Sound reference area concentrations across the risk range of 1 × 10^{-6} to 1 × 10^{-4}, except for three sediment RBTCs. Sediment RBTCs for the child Tribal RME and the API RME scenarios (at 1 × 10^{-4} risk level) and for the protection of river otters are within the range or higher than the concentration range of samples collected from upstream of the LDW and from the Upper Turning Basin (subsurface sediment).
- No sediment RBTCs were derived for human seafood consumption scenarios for arsenic, cPAHs, and dioxins and furans. Sediment RBTCs for arsenic and cPAHs for human seafood consumption could not be estimated because the relationships between concentrations in clam tissue and sediment are not well understood and are highly uncertain. Sediment RBTCs for dioxins and furans for human seafood consumption could not be estimated because there were no dioxin/furan tissue data from the LDW when the risk assessment was conducted.
- For arsenic, sediment RBTCs for direct-contact scenarios at the 1 × 10^{-6} risk level are below the concentration range of samples collected from upstream of the LDW and from the Upper Turning Basin (subsurface sediment) and are just below or within the wider range of concentrations in Puget Sound reference area samples. The tribal clamming direct-contact sediment RBTC at the 1 × 10^{-5} risk level is above the upstream concentration and the same as from the Upper Turning Basin (subsurface sediment) concentration but within the range of concentrations in Puget Sound reference area samples. The beach play and netfishing direct-contact sediment RBTCs at the 1 × 10^{-5} risk level are above concentrations found in Puget Sound reference areas, upstream of the LDW, and the Upper Turning Basin. Direct-contact sediment RBTCs at the 1 × 10^{-4} risk level are above all of these concentrations.
- Sediment RBTCs for cPAHs for the tribal clamming and beach play direct-contact scenarios at the 1 × 10^{-6} risk level are within the range of concentrations in samples collected from Puget Sound reference areas,

upstream of the LDW, and the Upper Turning Basin (subsurface sediment). The netfishing sediment RBTC for cPAHs at the 1 × 10^{-6} risk level is above all of the aforementioned concentration ranges as are all direct-contact sediment RBTCs at the 1 × 10^{-5} and 1 × 10^{-4} risk levels.

- Sediment RBTCs for dioxins and furans for direct-contact scenarios across the excess cancer risk range of 1 × 10^{-6} to 1 × 10^{-4} are above the concentration ranges of samples collected from Puget Sound reference areas and upstream of the LDW.

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